A Life Cycle Assessment (LCA) on the retrieval and waste management of derelict fishing gear

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A Life Cycle Assessment (LCA) on the retrieval and waste management of derelict fishing gear

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Signed: F. Silveides

DECLARATION OF AUTHORSHIP

I am the author of this thesis. The work described therein was carried out by myself personally.

Signed: F. Silveides

Abstract

Marine litter is considered a common concern of humankind that poses a serious risk to marine life, human health and the economy. One of the most abundant and harmful types of marine litter is abandoned, lost or otherwise discarded fishing gear. As it comprises of durable materials like nylon or polyethylene it can continue to catch fish for years before it breaks down into smaller pieces. The smaller pieces can then be ingested with possibly fatal consequences. To mitigate those impacts, derelict fishing gear [DFG] is increasingly retrieved from the ocean. However, apart from landfilling there is currently no established waste treatment system to process this material in Europe.

In the context of the European Interregional project MARELITT Baltic and in collaboration with WWF Germany and PreZero, this thesis investigated the retrieval and alternative waste treatment options for DFG. In total, four scenarios, namely the (1) mechanical recycling, (2) chemical recycling, (3) energy recovery and (4) disposal were evaluated to assess their potential environmental impact and feasibility. For this, industrial experiments and a life cycle assessment (LCA) were conducted.

The experiments highlighted DFG as a challenging mix of materials. Apart from large bulky items such as anchors, chains and cables, also smaller contaminants like sediments, salt and lead were contained. While this made the pre-treatment very time-consuming, a technical feasibility of the recycling techniques could be shown. The LCA results indicate that the mechanical recycling and energy recovery achieve the lowest potential environmental impacts. The chemical recycling scenario was too energy intensive to be environmentally competitive. Due to the determined lead content of up to 13.5% by weight in DFG, a disposal should be avoided as it poses a significant potential impact to human toxicity.

For the establishment of a waste treatment system and as far as possible, existing infrastructure such as local incineration plants or recycling facilities for end-of-life fishing gear should be used. As this requires the setup of a pre-treatment process, harbour personnel or other stakeholders should be encouraged to conduct those tasks. Further work is required to include social, economic and possibly other aspects, before a decision on the most appropriate waste treatment system can be made. Still, given DFG's harmful nature and challenging composition, preventive measures like the introduction of an extended producer responsibility scheme for fishing gear are key.

Acknowledgements

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Firstly, I would like to thank my supervisors at the University of Bath, Prof Marcelle McManus, Dr Sophie Parsons and Dr Sally Clift from the Department of Mechanical Engineering. They provided me with invaluable confidence, constructive critique and editorial support. Thank you for your patience, flexibility and shared expertise.

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List of Research Outputs

Authors (year)	**Title**	**Type**	**Short description:**
Schneider et al. (2018)	Collected marine litter - A growing waste challenge	Journal article (Marine Pollution Bulletin)	Systematic literature review on the collection and waste treatment of marine litter
Schneider (2018)	A novel material recycling pathway for ghost fishing gear – lessons from Germany	Oral presentation (6th International Marine Debris Conference)	Presentation of technical experiments, process challenges and development needs
Schneider (2018)	Life cycle assessment of retrieved ghost nets from the Baltic Sea	Oral presentation (MARELITT Baltic Conference)	Presentation of preliminary LCA results
Stolte and Schneider (2018)	Recycling Options for Derelict Fishing Gear (DFG)	Technical Report for the European Union	Description of DFG recycling experiments that were conducted as part of MARELITT Baltic
Schneider et al. (2020)*	Life cycle assessment on waste management options for derelict fishing gear*	Journal article (International Journal of Life Cycle Assessment)*	(* = Planned) LCA study to compare DFG waste treatment scenarios

List of Abbreviations

APC	Air pollution control
APt	Terrestrial acidification potential
Aquafil	Aquafil Global
BOF	Basic oxygen furnace
BREF	Best Available Techniques reference documents
BS	British Standard
CC	Climate Change
CETMAR	Centro Tecnológico del Mar
DB	dichlorobenzene
DEFRA	Department for Environment, Food & Rural Affairs
DFG	Derelict fishing gear
EAF	Electric arc furnace
EC	European Commission
EEA	European Environment Agency
EMEP	European Monitoring and Evaluation Programme
EN	European Standard
EOLFG	End-of-life fishing gear
EPfw	Freshwater eutrophication
EPm	Marine eutrophication
ETfw	Freshwater ecotoxicity
ETm	Marine ecotoxicity
ETt	Terrestrial ecotoxicity
EU	European Union
FAO	Food and Agriculture Organization of the United Nations
FD	Fossil depletion
FU	Functional Unit
GWP	Global warming potential
HT	Human toxicity
ILCD	International Reference Life Cycle Data System
ISO	International Organisation for Standardisation
LCA	Life cycle assessment
LCI	Life cycle inventory
LFG	Landfill gas
LOI	Loss on ignition
MAKSC	Magdeburger Kunststoff-Service-Center
MARELITT	Marine Litter
MARPOL	International Convention for the Prevention of Marine Pollution from Ships
MD	Metal depletion
MSW	Municipal solid waste

NMVOC	Non-methane volatile organic compounds
NOAA	National Oceanic and Atmospheric Administration
Nofir	Norsk Fiskeriretur
OD	Ozone depletion
PA	Nylon
PCU	Plastic cleaning unit
PE	Polyethylene
PEF	Product Environmental Footprint
PET	Polyethylene terephthalate
Plastix	Plastix Global
PM	Particulate matter formation
POF	Photochemical oxidant formation
PP	Polypropylene
PS	Polystyrene
RWTH	Rheinisch-Westfälische Technische Hochschule
SETAC	Society of Environmental Toxicology and Chemistry
TOC	Total organic carbon
UNEP	United Nations Environment Programme
US/USA	United States of America
US EPA	United States Environmental Protection Agency
WD	Water depletion
WSU	Water separation unit
WWF	World Wildlife Fund for Nature

1 Introduction

1.1 Marine litter

Human actions have caused several environmental shifts that reduce the biological diversity of our planet. This is not only true for overexploitation, land reclamation and climate change, but also for the dumping of waste (Derraik, 2002). Possibly the biggest global waste sink is the ocean. It accumulates different types of anthropogenic waste, called marine litter or marine debris. This includes glass, metals, wood, rubber, textiles and paper. However, its biggest and presumably most harmful contributor are plastics (Galgani et al., 2019).

Plastics are extremely durable, lightweight materials that are inexpensive to produce. This makes them the obvious choice for numerous applications and explains their growing production rates (Plastics Europe, 2019). Yet, from the estimated 8,300 million tonnes of plastics ever produced, approximately 6,300 million tonnes already became waste (Geyer, Jambeck and Law, 2017). While roughly 21% of this were incinerated or recycled, 79% were landfilled or left in the environment (Geyer, Jambeck and Law, 2017). To become marine litter, two major pathways exist. Land-based waste including from littering and landfills can travel by wind, rivers and wastewater, whereas waste from ocean-based activities such as fishing, shipping or aquaculture can directly enter the sea. At an estimated daily input of 12.9 to 35.1 thousand tonnes of mismanaged plastic waste (Jambeck et al., 2015), land-based sources account for approximately 80% of all marine litter today (Li, Tse and Fok, 2016). The remaining 20% from ocean-based activities mainly comprise of fishing gear.

Inside the ocean, high-density marine litter sinks to the seafloor whereas lighter materials can stay in the water column. Ocean currents can transport it to hot spot areas including underwater obstacles, beaches and ocean gyres (United Nations Environment Programme [UNEP], 2016). This cannot only cause smothering at the seafloor (Kühn, Bravo Rebolledo and van Franeker, 2015) but also navigation threats (Cho, 2005). The movement of marine litter also facilitates the spread of attached invasive species (Kiessling, Gutow and Thiel, 2015) and increases the burden to tourism (Keswani et al., 2016). Most notably however, it can lead to harmful interactions with marine life. In fact, entanglements and ingestions are known to affect more than 395 marine species worldwide (Gall and Thompson, 2015). Particularly after time, when larger plastics break down into microplastics, the risk of ingestion increases for smaller species (i.e. Gutow et al., 2019). As microplastics can adsorb chemicals, there is additional concern about a possible bio-accumulation into the human food-chain (Ribeiro et al., 2019).

The Sustainable Development Goals of the United Nations call to "prevent and significantly reduce marine pollution" by 2025 (United Nations General Assembly, 2015). To achieve this, a transition from the current linear economy towards a more circular economy is proposed (Lieder and Rashid, 2016). At its core, it means the introduction of closed-loop material flows and to encourage prevention, reuse and recycling over recovery and disposal techniques (Lieder and Rashid, 2016). Especially in the European Union, the implementation of a circular economy has been a key task since the adoption of its action plan in 2015 (European Commission, 2015). As a result, more ambitious recycling targets were introduced (i.e. Directive (EU) 2018/852, 2018) and specific proposals on fishing gear and single-use plastic items were made (European Commission, 2018). However, while preventive measures address marine litter at its source and decrease the waste input in the long-term, they do not reduce the current level of pollution.

The approaches to reduce existing marine litter include beach clean-ups, mid-ocean collections and seafloor retrievals (Schneider et al., 2018). In all cases, the collected marine litter is brought to land and given to a waste treatment. While detailed information on the compositions of marine litter is rare (Schneider et al., 2018), it can be expected to be contaminated with salt and sand and that a degradation processes has already taken place. This reduces its chance for a reuse, recycling and recovery, making landfilling an even more likely treatment option. As poorly managed landfills are major contributors to marine litter themselves (Jambeck et al., 2015), it is possible that previously collected marine litter starts its journey from landfills back to the sea. Even for well-managed landfills there is serious concern about its environmental impacts. In other words, there is a need to identify suitable pathways for marine litter when it comes out of the sea.

1.2 Life Cycle Assessment

Life cycle assessment [LCA] is a method for calculating the potential environmental impacts of product systems over their life cycle (International Organisation for Standardisation [ISO], 2018). It is a holistic approach that considers multiple environmental impact categories at the same time (European Commission, 2011). This allows LCA to compare alternative product systems, to identify hot spot areas and to investigate when burden shifts between impact categories, life cycle stages or regions occur (Finnveden et al., 2009). Although LCAs can be data and time intensive, they allow for better-informed decisions to take place (Parsons et al., 2019).

Originally applied to beverage containers, LCA quickly found its way to the chemical industry who promoted the tool worldwide (McManus and Taylor, 2015). Today, LCA is used by different stakeholders including companies and governments to improve products and guide their strategic planning (McManus et al., 2015). In the area of waste management, it is frequently applied to compare alternative waste treatment techniques (Laurent et al., 2014a).

LCAs include the following steps: (1) the goal and scope definition, (2) the compilation and analysis of the life cycle inventory, (3) the translation of the life cycle inventory into potential environmental impacts, and (4) the result interpretation (ISO, 2018; ISO 2006). Although countless methodological challenges exist (McManus et al., 2015), LCA is built on a strong methodological foundation (Finnveden et al., 2009) which makes it one of the most accepted tools to determine the environmental performance of systems today. LCA is therefore a suitable tool to help determine appropriate solutions for the waste management of marine litter.

1.3 Aims and objectives

This thesis identifies the most environmentally sustainable waste management option for derelict fishing gear (DFG). DFG was selected as a case study, because it is one of the most abundant and harmful types of marine litter. The individual research objectives are provided below:

Objective 1: Illustrate the current knowledge on DFG waste management options

a) Map DFG retrieval methods in specific locations
b) Identify waste treatment pathways for DFG
c) Review relevant LCA studies from the literature

Objective 2: Investigate the technical feasibility of DFG waste management options

a) Establish a process flow chart for DFG recycling, recovery and disposal pathways
b) Present original experimental data for DFG retrieval and recycling trials
c) Critically evaluate the pre-treatment, recycling and recovery challenges
d) Compile a comprehensive waste composition for DFG

Objective 3: Establish the potential environmental impacts for DFG management options

a) Provide life cycle inventory data for novel recycling processes used to process DFG
b) Identify environmentally significant recycling, energy recovery and disposal processes
c) Investigate the sensitivity of relevant input parameters on the results

1.4 Thesis structure

To address the research objectives, this thesis is divided into 12 Chapters (Figure 1.1). This introductory Chapter is followed by a literature review on marine litter and life cycle assessment in Chapter 2 and 3. The methodology is introduced in Chapter 4, before information about the conducted experiments and the life cycle inventory is provided in Chapter 5-9. Chapter 10 describes the results of the life cycle impact assessment. Ultimately, the thesis ends with a discussion and conclusion in Chapter 11 and 12 (Figure 1.1).

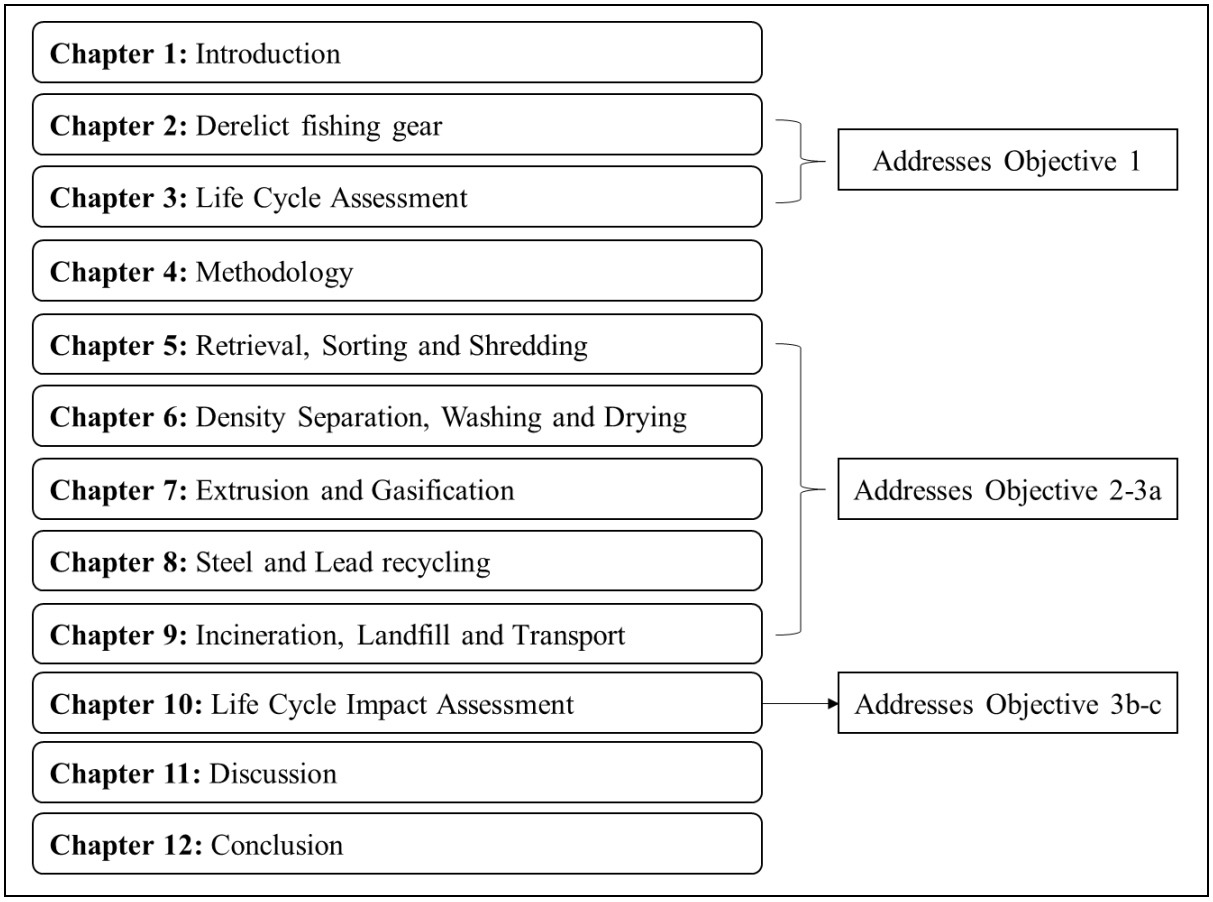


Figure 1.1: Thesis structure and addressed research objectives

1.5 Research scope

The research links the areas of marine litter, waste management and sustainability assessment. However, as those are very broad areas a clear research scope needs to be defined. Within marine litter, only derelict fishing gear from the seafloor is considered. From the area of waste management, only recycling, recovery and disposal techniques are evaluated and as part of a sustainability assessment only the environmental impacts are modelled (Table 1.1).

Table 1.1: Definition of the research scope

Area	Sub-area	Included	Excluded
Marine litter / Derelict fishing Gear	From seafloor	x	
	From beaches		x
	From sea surface		x
Waste Management Pathways	Disposal	x	
	Energy Recovery	x	
	Chemical Recycling	x	
	Mechanical Recycling	x	
	Reuse		x
	Prevention		x
Sustainability Assessment	Environmental impacts	x	
	Economic impacts		x
	Social impacts		x

The currently excluded aspects such as marine litter from other ocean compartments, the technical and environmental assessment of reuse and prevention pathways as well as the potential economic and social impacts provide scope for future research.

1.6 External contributions

The research in this thesis is the result of a collaboration between the University of Bath, the World Wildlife Fund for Nature [WWF] Germany and the waste management company PreZero (previously Toensmeier). While the University of Bath funded the research and provided continuous support, WWF Germany and PreZero established the industrial contacts and financed the experiments described in this thesis. An overview of the research partners and their contributions is provided in Table 1.2.

Table 1.2: Overview of external and internal contributions

<table>
<tr><th>Experiments</th><th>Experiment Setup</th><th>Data collection</th><th>Data compilation / LCA</th></tr>
<tr><td>Retrieval</td><td colspan="2">WWF Germany + fishers</td><td rowspan="9">Author's own contribution</td></tr>
<tr><td>Sorting</td><td rowspan="3">Vecoplan</td><td rowspan="3">Author's own contribution</td></tr>
<tr><td>Shredding</td></tr>
<tr><td>Washing</td></tr>
<tr><td>Density Separation</td><td>Vecoplan, Andritz Separation</td></tr>
<tr><td>Drying</td><td colspan="2" rowspan="2">MAKSC, HS Magdeburg</td></tr>
<tr><td>Extrusion</td></tr>
<tr><td>Gasification</td><td>CleanCarbonConversion</td><td>Author's own contribution</td></tr>
</table>

WWF Germany oversaw the retrievals (Table 1.2) and the overall project management for the waste treatment options. This was part of a European funded Interregional project called MARELITT Baltic. Carried out between 2016 and 2019 with partners from Sweden, Poland and Estonia, this project had the aim to generate and share best practice knowledge for the handling

of derelict fishing gear. Although this thesis was not directly part of the EU project, it provided the research context and access to a wider audience. The author of this thesis accompanied WWF Germany on a retrieval trip near Freest to better understand the process.

The key industrial partners that have contributed to the research presented in this thesis were Vecoplan, Andritz Separation and CleanCarbonConversion. U. Kramer, A. Grose and S. Kolodzey from Vecoplan conducted the sorting, shredding, density separation and washing experiments (Table 1.2). A. Siebelitz and his colleagues from Andritz Separation tested the density separation (Table 1.2) and F. Rupert, M. Haupt and M. Küttel from CleanCarbonConversion processed DFG in their steam gasification plant (Table 1.2). The author of this thesis was present during all experiments to help the material handling, to take weight measurements and to suggest process improvements.

The main research partners that have contributed to this research were the Magdeburger Kunststoff-Service-Center [MAKSC] and Prof. G. Gerke's Circular Economy Department at the Hochschule Magdeburg. They investigated the mechanical recycling via a drying and extrusion (Table 1.2). The author of this thesis was not present during the experiments but gathered data via face-to-face meetings and phone conversations.

Several other companies like EREMA and Plastix Global also provided reference information and analysis results. While all external contributions are highlighted throughout the thesis, their importance for this thesis is kindly acknowledged here.

2 Derelict fishing gear

2.1 Research context

2.1.1 Rationale for fishing gear

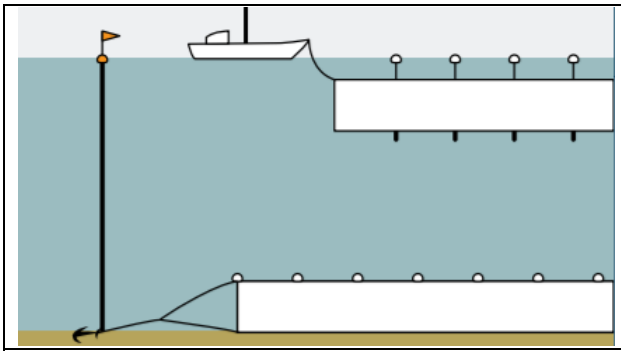
Marine litter comprises of a diverse mix of materials, particularly plastic waste. Because of its significant quantity and high impact in the ocean, the thesis focuses on abandoned, lost or otherwise discarded fishing gear, also called derelict fishing gear [DFG].

2.1.2 Types of fishing gear

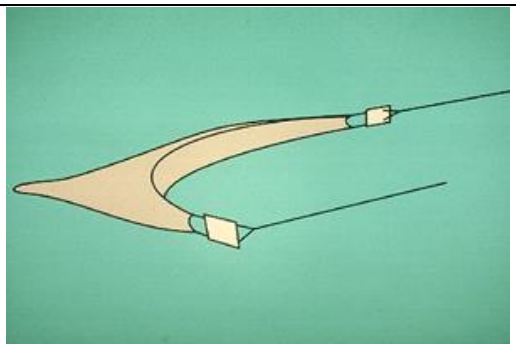
Fishing is an important economic activity that provides food, jobs and economic growth, and depending on the target species, different types of fishing gear are used. Those include nets, traps and lines as well as fish aggregating devices (Richardson, Hardesty and Wilcox, 2019). Nets can be further divided into seine nets, trawl nets and gillnets (Ibid). From those, gillnets and trawl nets are described in more detail, because they were found in retrieval experiments by WWF Germany (Chapter 5.1.1).

Gillnets

Gillnets are vertical walls of netting positioned in the water column or at the seafloor to enmesh fish within (Food and Agriculture Organization of the United Nations [FAO], 2020a; Figure 2.1). To keep their vertical orientation, floats are attached to the upper line and weights to the bottom line (FAO, 2020a). When positioned at the seafloor, anchors and buoys are used to fix and indicate their position (FAO, 2020a). The netting, floaters and buoys are typically made of nylon [PA], polyethylene [PE] or polypropylene [PP], while the weights and anchors usually comprise of lead and steel respectively. Compared to other types of fishing gear, gillnets are most frequently lost and pose the highest threat to marine life (Huntington, 2016).



Gillnets
Image adapted from: Huntington, 2016



Trawl net
Image adapted from: FAO, 2020b

Figure 2.1: Schemata of trawl and gillnets in operation

Trawl nets

Trawl nets are towed behind a ship so that fish can be caught in its cone-shaped netting (FAO, 2020b; Figure 2.1). A headline and a ground rope mark the opening of the trawl (FAO, 2020b). To avoid damages when the trawling takes place at the seafloor, rock hoppers can be attached to the ground rope (FAO, 2020b). The netting and ropes mainly comprise of PA, PE and PP whereas rock hoppers are typically made of rubber. Trawl nets are described as the least commonly lost and least harmful type of fishing gear (Huntington, 2016).

2.1.3 Derelict fishing gear and its impacts

Fishing gear is commonly lost through severe weather conditions, gear conflicts and underwater obstacles (Richardson, Hardesty and Wilcox, 2019), while human errors, vandalism and illegal disposals also take place (Deshpande et al., 2020). The resulting annual input of fishing gear into the ocean is estimated at 640 thousand tonnes for the world (MacFayden, Huntington and Cappell, 2009), at 380 tonnes for Norway (Deshpande et al., 2020) and at 1000 tonnes for the Baltic Sea (Bertling and Nühlen, 2019). This makes DFG one of the biggest single sources of marine litter with an overall contribution of approximately 10% (MacFayden, Huntington and Cappell, 2009). The remaining sea-based litter mainly occurs as losses from ships, oil-platforms or other sea-based infrastructure in form of containers, protective gear or other items.

Designed to catch fish, fishing gear can continue to entangle marine life after becoming derelict, a process called ghost fishing. As such, it cannot only cause cuts and slow down animals decreasing their ability to escape from threats, but also result in drowning and starvation (Nelms et al., 2016). DFG entanglements are particularly common for large taxa like whales, dolphins, turtles and fish (Gall and Thompson, 2015). For example, 80% of right whales and 50% of humpback whales showed marks of entanglement (Knowlton et al., 2012; Robbins and Mattila, 2004), while a study on sealions revealed that younger and more curious animals can have higher incidents rates (Lawson et al., 2015).

Another important impact of DFG is smothering, a term used to describe the entanglement of flora and fauna at the seafloor (Kühn, Bravo Rebolledo and van Franeker, 2015). Smothering cannot only reduce the light penetration and thus prevent plant growth, but also suffocate corals when pushing them into anoxic sediments (Valderrama Ballesteros, Matthews and Hoeksema, 2018). Once the corals die, DFG can harm marine life while dragging along the seafloor towards the next resting spot where it can start the cycle again (Kühn, Bravo Rebolledo and van Franeker, 2015).

As briefly described in Chapter 1.1, there are other marine litter impacts which also apply to DFG. Those include ingestions (Gall and Thompson, 2015), navigation obstacles (Cho, 2005) or economic damages to fishing, shipping or tourism (Scheld, Bilkovic and Havens, 2016). While the ingestion of fishing gear components can also be lethal (Franson et al., 2003; Zabka et al., 2006), it is the risk for entanglements that make DFG the most harmful type of marine litter today (Wilcox et al., 2016).

2.1.4 Input prevention

To prevent marine litter and derelict fishing gear, multiple legislative and industrial actions have been taken. Some of the most relevant developments for Europe are briefly introduced.

Legislative developments

In 1988, MARPOL Annex V came into force. Apart from banning the discharge of plastics from ships in general, it also introduced specific provisions for fishing gear. For example, lost or discharged fishing gear must be recorded and reported to the flag state, including information on the location, gear type, size, quantity, material composition and buoyancy (Marine Environment Protection Committee, 2017).

The Council Regulation (EC) No 1224/2009 (2009) brings those MARPOL Annex V requirements into European law. It also requires fishing vessels to carry retrieval equipment on board and to attempt a retrieval of fishing gear as soon as possible after it has been lost. A failure to report lost fishing gear can result in the imposition of retrieval costs on the master of the fishing vessel. Despite this, fishing gear losses are not commonly reported which indicates that a law enforcement is difficult to achieve.

Another relevant regulation for fishing gear is the European Directive on Port Reception Facilities (Directive (EU) 2019/883, 2019) that requires waste from ships to be landed and adequately managed in ports. While vessel owners were traditionally asked to directly pay for the landed waste, the current legislation proposes an indirect fee. If implemented, this would remove an incentive for littering, as no immediately payable cost for waste occurs. In addition, an extended producer responsibility scheme for fishing gear was recently proposed (European Commission, 2018). If adopted, the collection and treatment cost for fishing gear would then be paid by the fishing gear producer. This would not only incentivise the gear producer to design fishing gear to be recyclable but also to setup an effective return system. Until those proposals are implemented, unnoticed gear losses will continue to take place.

Market developments

Since 2008, three major companies have entered and dominated the European market for the collection and recycling of end-of-life fishing gear [EOLFG], namely Norsk Fiskeriretur [Nofir], Aquafil Global [Aquafil] and Plastix Global [Plastix].

Nofir was set up in 2008 to develop a collection and recycling system for fishing gear in Norway (Nofir, n.d.). Today, they are widespread across Europe, Africa, Asia and America collecting and treating between 4.7 to 7.4 thousand tonnes of fishing gear per year (Ibid). While Nofir focuses on the collection, transport and dismantling, the recycling is mainly carried out by Aquafil. Nofir's dismantling plants are based in Lithuania and Turkey.

Aquafil operates a chemical recycling plant in Slovenia where polyamide rich waste such as carpets or fishing nets are turned into "Econyl" yarn (Aquafil, n.d.). Although there is not much transparency about the quantity or type of fishing gear that gets recycled, it is generally understood that less contaminated end-of-life fishing gear is used (Figure 2.2).

Plastix is based in Denmark where it produces pellets from PP and PE fishing nets via a mechanical recycling. After EOLFG arrives at its facility, the production process involves a dismantling, shredding and density separation as well as a washing, drying and extrusion. Still, like Aquafil, only the less contaminated EOLFG is accepted as an input material (Figure 2.2).

		
Nofir – Image adapted from: Nofir, 2017	Aquafil – Image adapted from: Aquafil, 2018a	Plastix -Image adapted from: Clean Nordic Oceans, n.d.

Figure 2.2: End-of-life fishing gear input for major European recycling companies

Other European companies that recycle EOLFG are Verdura, Ecoalf and Klattermusen (Charter, Carruthers and Jensen, 2018). While they turn EOLFG directly into backpacks, clothing or shoes, Aqufil and Plastix sell their yarn and pellets as raw materials for example to Adidas or Interface on a larger scale. The start-up company Fishy Filaments produces raw materials for 3D-printing applications but not yet at a larger scale.

Link to derelict fishing gear

An increased collection and recycling as well as the legislative developments above can prevent EOLFG to become DFG. This makes them more cost effective and efficient compared to removal techniques (Ryan et al. 2009). However, a prevention does not reduce the current stock of DFG, and a full stop of losses would be optimistic to assume. To cure existing impacts, retrievals are needed alongside preventive techniques.

2.2 Marine litter collections

2.2.1 Overview

As part of this thesis a systematic literature review on marine litter collections was conducted (Schneider et al., 2018). In the review scientific and non-scientific marine litter collections were distinguished. Scientific collections followed clear sampling protocols to monitor marine litter but did not reduce marine litter at scale. Non-scientific collections on the other hand, were less explicit about the methods they used while cleaning-up large quantities. In general, marine litter removal efforts appeared to have intensified over time, as significantly more collections were reported after 2009 (Ibid). They were carried out worldwide, especially around North America, Brazil, Australia and Europe (Figure 2.3).

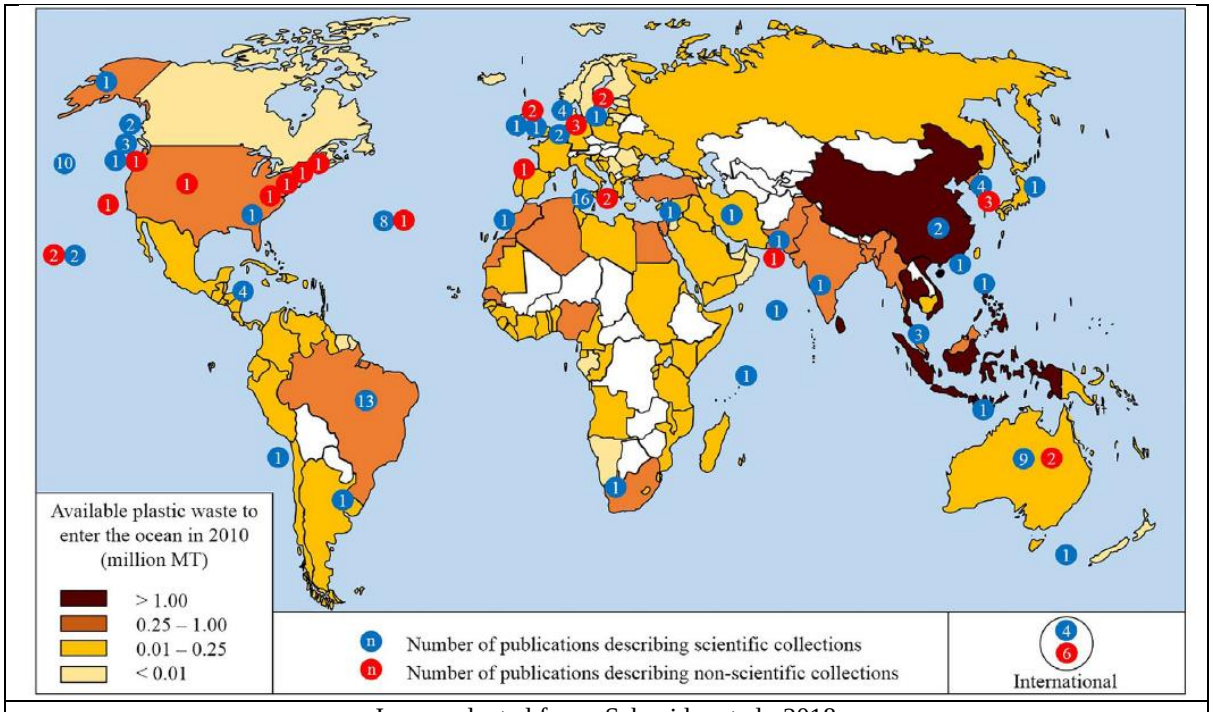


Image adapted from: Schneider et al., 2018

Figure 2.3: Map of global marine litter collections

In total, 16 collection methods were identified from which nine were solely used in small-scale scientific collections (Schneider et al., 2018). The remaining methods which were also used at a larger scale were manual beach clean-ups, retention booms and crane excavations as well as surface trawling, bottom trawling and diving with or without ROVs (Figure 2.4). From those, bottom trawling and diving focused specifically on DFG.

			
Beach clean-ups: Marine litter is removed from coastlines, usually manually although heavy machinery is sometimes employed as well	**Surface trawling or bottom trawling:** Retrieval gear grabs marine litter at the surface or seafloor, tows it behind the ship to be hived on board or landed onshore	**Retention booms:** Floating barriers that accumulate marine debris **Crane excavations:** Ship mounted cranes to remove marine litter from the surface or seafloor	**Diving (+ROVs):** Divers free and remove marine litter from shallow waters, diving can also be combined with ROVs
Image adapted from: GhostNets Australia, n.d.	Image adapted from: Sea Shepperd UK, n.d.	Image adapted from: Jung et al., 2010	Image adapted from: Becatoros, 2018

Figure 2.4: Common large-scale marine litter clean-up approaches

2.2.2 Retrievals

While the terms retrieval, removal and collection are used interchangeably, this thesis adopts the term retrieval to describe marine litter collections from the seafloor with a focus on DFG. Although retrievals are expensive and time-consuming (Williams and Rangel-Buitrago, 2019) they can be particularly beneficial to cure negative impacts. For example, after DFG retrievals in Puget Sound, previously displaced kelp and eelgrass showed a seasonal recovery rate of 100 and 30% respectively (June and Antonelis, 2009). Meanwhile the removal of 34,408 derelict pots over six years in the Chesapeake Bay increased the harvest of blue crabs by 27% (Scheld, Bilkovic and Havens, 2016). Ultimately, retrievals can also increase awareness and possibly result in a positive behaviour change (Rayon-Viña et al., 2019).

Large-scale retrieval projects

Examples of large-scale retrieval projects from the literature review (Schneider et al., 2018) are summarised in Table 2.1. Divers were mainly employed by the Olive Ridley Project, Healthy Seas and the Italian Institute of Marine Science [ISMAR] as well as by the National Oceanic and Atmospheric Administration [NOAA] of the United States of America [US/USA] and the Californian Fishing gear recovery project. The remaining initiatives mainly applied bottom trawling as retrieval technique. Derelict pots and traps were commonly retrieved across

the North American coastal states, whereas derelict nets occurred more regularly during operations in Europe, Asia and Hawaii. The two largest retrieval projects for derelict nets removed over 800 tonnes from Hawaii and almost 300 tonnes from the Baltic Sea (Table 2.1).

Another important large-scale retrieval is the annual survey for DFG in Norway. Partially funded by the Government and fishers, this initiative removed over 1000 tonnes of DFG since its foundation in the 1970s (G. Langedal, personal communication, 21 March 2019).

Table 2.1: Large-scale DFG retrieval projects

Region		**Initiator**	**Main Method**	**Year**	**DFG Removal**	
Asia	Maldives	Olive Ridley Project	Diving	2013-2016	1	t
North America	Hawaii	NOAA	Diving	1996-2014	820	t
	Washington	Northwest Straits foundation	Diving	2002-2016	37+	t
	New Brunswick	Fundy North Fisherman's Association	Bottom trawling	2008-2015	2+	t
	Maine	Gear grab	Bottom trawling	2000-2006	5,600	pcs
	Virginia	CCRM: VIMS	Bottom trawling	2008-2013	33,297	pcs
	California	Fishing gear recovery project	Diving	2006-2012	60	t
Europe	Poland	WWF Poland	Bottom trawling	2015	268	t
	Spain	CETMAR	Bottom trawling	2009-2010	15.2	t
	Italy	ISMAR	Diving	2014-2016	0.5	t
	Multiple	Healthy seas	Diving	2000-2006	20	t

Table adapted from: Schneider et al. (2018)

Limitations

For retrievals to be effective, they require information about the location of DFG. If not reported by fishers, divers or federal authorities (Valderrama Ballesteros, Matthews and Hoeksema, 2018; Donohue et al., 2001), underwater cameras or side-scan sonars can be used (Spirkovski et al., 2019) adding additional cost to the retrieval process.

Retrievals cannot be conducted everywhere. For example, in sensitive areas such as munition dumpsites, additional thoughts to legal and safety implications need to be given (Sahlin and Tjensvoll, 2018). In addition, while bottom trawling can be carried out in great depth, diving operations are restricted to shallow waters (Stelfox, Hudgins and Sweet, 2016).

There is also concern about the environmental impact of the retrieval operation (Ryan et al., 2009). For example, when inactive gear is found in in reef habitats, a retrieval may be more damaging than beneficial and not considered at all (Sahlin and Tjensvoll, 2018).

2.2.3 Waste composition

There is very little information on the composition of retrieved DFG. The Northwest Straits Foundation (2015) report that from 37 tonnes of DFG, “22.9 tonnes [61.9%] of mostly leadline

were recycled", while WWF Poland (2015) suggest that from 268 tonnes of DFG, 77.6% comprised of nets and ropes, 15.7% of steel, 3.9% of mixed waste, 2.5% of cables and 0.3% of plastic boxes. The 15.2 tonnes (45%) of DFG retrieved by the Spanish Centro Tecnológico del Mar [CETMAR] (Table 2.1), were accompanied by 6.9 tonnes (20%) other waste, 4.1 tonnes (12%) metals and 3.7 tonnes (11%) mixed plastics as well as 1.8 tonnes (5%) textiles, 1.2 tonnes (4%) rubber and 0.9 tonnes (3%) timber (Lueiro, 2015).

Although not commonly reported, a notable amount of sediments, organic debris and salt can be expected to attach to the waste fractions. This adds to the very diverse mix of materials in DFG (also see Figure 2.5) making it difficult to find appropriate waste treatment techniques. In fact, the Olive Ridley Project (n.d.) describes it as "One of the greatest obstacles [… to identify] what to do with the vast quantity of ghost gear collected.".

Image adaptded from: Healthy Seas, 2018

Image adaptded from: Healthy Seas, 2019

Figure 2.5: Retrieved derelict fishing gear

2.3 Waste treatment

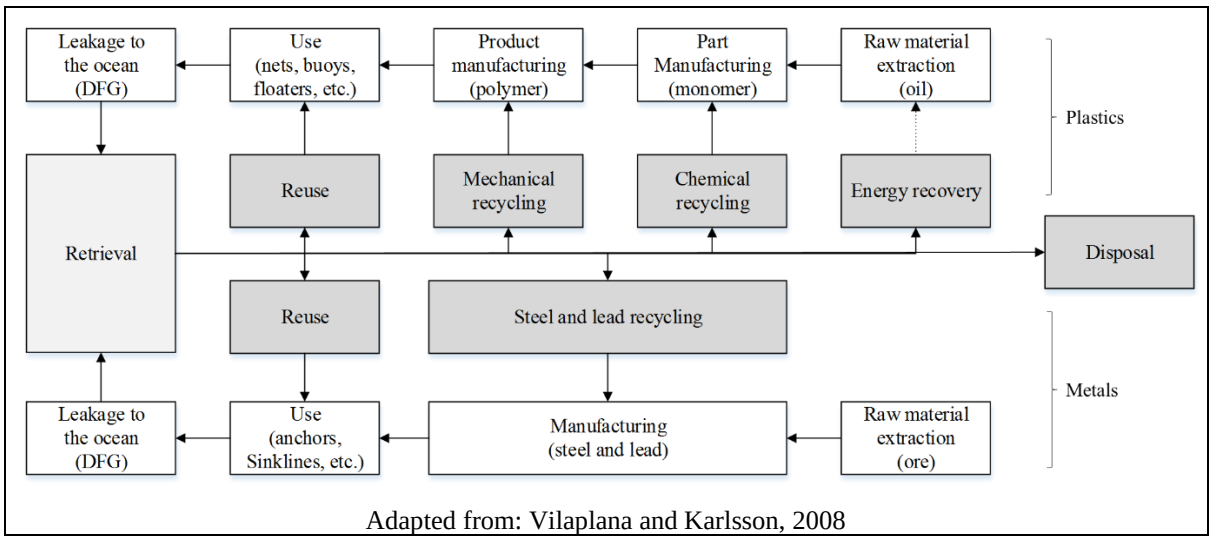
2.3.1 Overview

Waste treatment options can be divided based on the waste hierarchy into a reuse, recycling and recovery as well as a disposal. While their exact legal definition can be found elsewhere (Directive 2008/98/EC, 2008), a brief summary is given here:

- **Reuse**: checking, cleaning and repairing waste to fulfil its original function.
- **Recycling**: reprocessing waste into new products except for fuels and backfilling.
- **Recovery**: reprocessing waste to substitute otherwise needed materials; referred to as energy recovery when fuels are replaced.
- **Disposal**: storing waste on a long-term without a primary material substitution.

Fishing gear mainly comprises of plastics and metals. A reuse, recycling and disposal are applicable to both material groups while a recovery is only suitable for the high calorific plastic

waste. The recycling of plastics can be further divided into a mechanical and chemical recycling depending on whether the chemical structure of the polymers is changed (Al-Salem, Lettieri and Baeyens, 2009). The difference between those waste treatment options is the increasing length of the loop back to the original life cycle (Figure 2.6).



Adapted from: Vilaplana and Karlsson, 2008

Figure 2.6: Generic waste treatment options for derelict fishing gear

All waste treatment options can require several pre-treatment steps. In addition, a disposal, is typically carried out in a landfill, an energy recovery via incineration and a mechanical recycling via extrusion. A chemical recycling is usually achieved through a pyrolysis or gasification process whereas a metal recycling is commonly conducted in pyrometallurgical furnaces.

To identify examples of DFG waste treatments, a literature review using Scopus with the key words “marine litter”, “marine debris” or “fishing gear” in combination with “reuse”, “recycling”, “recovery”, “disposal”, “landfill”, “incineration”, “pyrolysis”, “gasification” or “extrusion” was conducted in December 2019. At the same time, information about the waste treatment of DFG from personal discussions, conferences and large-scale retrieval projects were included. The latter was obtained from the respective websites of the initiatives mentioned in Table 2.1. The results are summarised in the sub-chapters below. However, as new pathways for DFG are frequently emerging, the presented information should not be viewed as a set list.

2.3.2 Reuse

The reuse value of DFG is generally very low and apart from undamaged pots, traps and ropes (Fundy North Fisherman's Association, 2016; Northwest Straits Foundation, 2019) evidence for a reuse could not be found.

While a reuse typically implies to restore a product to its original function, other waste treatments with minimal reprocessing effort are included here as well. For example, ropes have been turned into bracelets, mats and art (Charter, Carruthers and Jensen, 2018); nets have been used in fencing, as soccer nets or to harden coastal tracks (MacFayden, Huntington and Cappell, 2009) and fibres were used to reinforce mortar (Orasutthikul, Unno and Yokota, 2017). Those approaches may be good to raise awareness, but they only recover specific components at a small-scale.

2.3.3 Recycling

As previously identified (Figure 2.6) the recycling of DFG can be divided into a mechanical, chemical and metal recycling. In this section examples are briefly introduced.

Mechanical recycling

In South Korea, DFG is retrieved, pre-treated and recycled as part of a National Marine Debris Management Plan (Jung et al., 2010). The pre-treatment includes a sorting, cutting and lead separation as well as a crushing, washing and drying process. Particularly the lead separation and washing are highlighted as important steps, as they remove hazardous materials, salt and sediments to improve the properties of the output material. After the pre-treatment, the plastics are extruded and used as a solid fuel for an energy recovery, while a use as raw material for the recycling sector is considered as well (Ibid).

Adidas received 75km of DFG from Sea Shepherd who retrieved it shortly after its disposal near the coast of Africa (K. George, personal communication, 30 January 2018). A fraction of the material was then cleaned, extruded into fibres and used to decorate the upper of a limited shoe edition (Ibid). This was possible due to the material's high initial purity and quality, but Adidas' attempts to turn more contaminated DFG into fibres have not been successful so far.

At a laboratory scale, DFG was pre-treated and extruded into test specimen to study its mechanical properties (Gerke et al., 2016). The pre-treatment included a manual sorting and several shredding steps before the extrusion took place. The results revealed that the mechanical properties of DFG were inferior to virgin material (Ibid).

While other companies such as Plastix Global in Denmark or Bureo in Chile are sometimes linked to the recycling of DFG, they tend to only accept very clean EOLFG. Large-scale initiatives that mechanically recycle contaminated DFG could not be found.

Chemical recycling

For marine litter, a pyrolysis and gasification have been proposed as an onboard treatment to power ship operations (Tunnicliffe, 2017; Panicker and Magid, 2016). In fact, floating marine debris from the Great Pacific Garbage Patch [an ocean gyre in the North Atlantic that accumulates marine debris comprising of more than 46% of DFG (Lebreton et al., 2018)], have already been turned into valuable fuel (Slat, 2014). Still, chemical recycling techniques have not yet been adopted for DFG at a large scale.

One exception may be Aquafil who claims to reprocess DFG into their Econyl yarn via a depolymerisation process (Aquafil, 2018b). For this Aquafil partnered with initiatives such as Healthy Seas and Nofir to retrieve and sort DFG prior to its own processing steps. However, it is not clear how contaminated DFG (Figure 2.5) can be turned into an acceptable input for the Aquafil plant (Figure 2.2). In case DFG can be used, its relatively small quantity in comparison to other nylon waste such as carpets or EOLFG in the Econyl yarn, seriously question the yarn's marketing claim as a ghost net product.

Metal recycling

Metals from DFG are recovered by Schnitzer Steel in the US as part of a fishing for energy initiative (McCoy, 2010), but it is not clear how or which metals are separated. Similarly, lead from DFG is sent to a recycling by the Northwest Straits Foundation (2015), but it is not clear how the lead was separated. In South Korea, lead is separated from PP ropes by air blowing (Jung et al., 2010), but it is not stated how the lead is treated afterwards. Still, as steel and lead recycling are established industries it can be expected that separated metal fractions will be adequately recycled.

2.3.4 Energy recovery

In South Korea, DFG is incinerated on small islands that lack other recycling facilities. The incineration process consists of a crushing, charging and burning as well as a steam raising and several air pollution control measures (Jung et al., 2010). The plants process up to 32 tonnes of marine litter per year meeting the national emissions standards (Ibid). Although Jung et al. (2010) mention the importance of a salt removal prior to incineration, it is not clear how this is incorporated into the process.

In North America and Hawaii DFG is commonly retrieved and reprocessed as part of the fishing for energy initiative. For this, harbours are supplied with lockable containers to collect the

material (McCoy, 2010). Although fishers are requested to provide the fishing gear in a dry and organic debris free condition, for convenience the material is typically not sorted (National Fish and Wildlife Foundation, n.d.). After the collection, full containers are shipped to Schnitzer Steel facilities for a cutting and metal separation before the remaining material is sent to Covanta incineration plants (McCoy, 2010). There the material is mixed with other waste and incinerated to recover electricity.

In Europe, DFG or similar marine litter is usually not accepted in conventional incineration plants, because its calorific value is too high for an efficient energy recovery (Fishing for Litter Scotland, 2015).

2.3.5 Disposal

Apart from North America and South Korea, where large-scale energy recovery routes for DFG exist (see above), landfilling can be expected to be the most common waste treatment approach. For example, the 268 tonnes of DFG that were retrieved from the Baltic Sea in 2015 (WWF Poland, 2015), needed to be landfilled due to a lack of other locally available waste treatment options. This is presumably also the case for most other initiatives that did not provide information on the DFG treatment (i.e. Lueiro, 2015).

2.4 Chapter summary

This chapter provided an overview of the fishing gear life cycle, covering its accumulation in the ocean, retrieval activities and subsequent waste treatment options.

More specifically, it was shown that large quantities of abandoned, lost or otherwise discarded fishing gear cause severe impacts in the ocean. To remove this hazard retrieval activities such as diving or dragging operations are increasingly conducted. While novel waste treatment solutions for landed DFG are frequently proposed, the chapter highlighted a lack of established DFG recycling and energy recovery schemes in Europe, indicating that most DFG is landfilled. It was also shown that quantitative data on the DFG waste composition and recycling processes is not revealed. However, as this is required to identify a suitable waste management system for the future, treatment processes need to be further assessed.

3 Life Cycle Assessment

3.1 Definition and historical development

Definition

Life cycle assessment (LCA) is a tool to assess the "potential environmental impacts […] throughout a product's life cycle from raw material acquisition through production, use, end-of-life treatment, recycling and final disposal" (ISO, 2018). Its holistic approach can avoid burden shifting not only between life cycle stages but also between different environmental problems and regions, making it one of the most suitable approaches to study environmental performances today (Finnveden et al., 2009).

LCA development - 1960s to 1980s

LCA has its origins in the 1960s and 1970s when increasing concern over finite resources and environmental pollution (McManus and Taylor, 2015) led to its development in North America and Europe at roughly the same time (Hunt and Franklin, 1996; Bousted, 1996). While possibly the first LCA was presented on the energy demand of different chemicals at the World Energy Conference in 1963 (Bjørn et al., 2018), most early work focused on packaging alternatives. For example, the Midwest Research Institute in the USA directed an internal study for Coca Cola in 1969 which provided the company with "comfort" to change their glass to plastic bottles (Hunt and Franklin, 1996); a study for the Mobil Chemical Company in 1972 which showed superiority of light weight polystyrene foam trays in comparison to heavier pulp trays (Hunt and Franklin, 1996); and a study for the US Environmental Protection Agency in 1974 that became the first peer-reviewed and publicly available LCA (Bjørn et al., 2018).

During the 1970s events like the publication of the "Limit to growth" (Meadows et al., 1972) or the oil crisis in 1973 and 1979 further increased the attention on resource scarcity. Still, while LCA work was adopted across some research institutions in Europe (Klöpffer, 1997; Oberbacher, Nikodem and Klöpffer, 1996; Fink, 1997; Hunt and Franklin, 1996), factors such as the lack of computers for the time-consuming inventory calculations (Bousted, 1996) prevented its wide-spread use. Only in the late 1980s when the first LCA software was released (Bjørn et al., 2018) and the treatment of solid waste became an issue, LCAs became widely adopted spreading into new domains (Klöpffer, 1997).

LCA development - 1990s to 2010s

The sudden uptake of LCA led to problematic marketing claims and in 1991 LCAs were consequently denounced in the USA until a consensus on the methodology was reached (United States Environmental Protection Agency [US EPA], 2006). To harmonise the LCA methodology, the Society of Environmental Toxicology and Chemistry [SETAC] established a working group in 1990 which coined the term life cycle assessment while producing several methodological reports (McManus and Taylor, 2015). Subsequently, the International Organisation for Standardisation developed a set of LCA standards which were published between 1997 and 2002. ISO 14040:2006 and ISO 14044:2018 form the current standard today.

During the standardisation process, LCAs methodological development did not stand still. For example, the introduction of impact assessment methodologies such as CML92 and Eco-indicator 99 in the 1990s (Bjørn et al., 2018) allowed LCA to evolve from inventory studies to an impact assessment technique (US EPA, 2006). This together with the publication of the Ecoinvent inventory database in 2003 further increased the uptake of LCA (Bjørn et al., 2018). However, as the initial set of data and standards were not yet mature enough for policy decisions, the UNEP/SETAC Life Cycle Initiative and the International Reference Life Cycle Data System [ILCD] were set up in the 2000s to provide for a more scientific approach (Bjørn et al., 2018). Among other outputs, this resulted in the ILCD handbook (European Commission, 2010) which represents one of the most comprehensive LCA guidelines today (Bjørn et al., 2018).

3.2 LCA methodology

3.2.1 Overview

LCAs are divided into four steps, namely the (1) Goal and scope definition, the (2) Inventory analysis, the (3) Impact assessment as well as the (4) Interpretation (ISO, 2006; Figure 3.1).

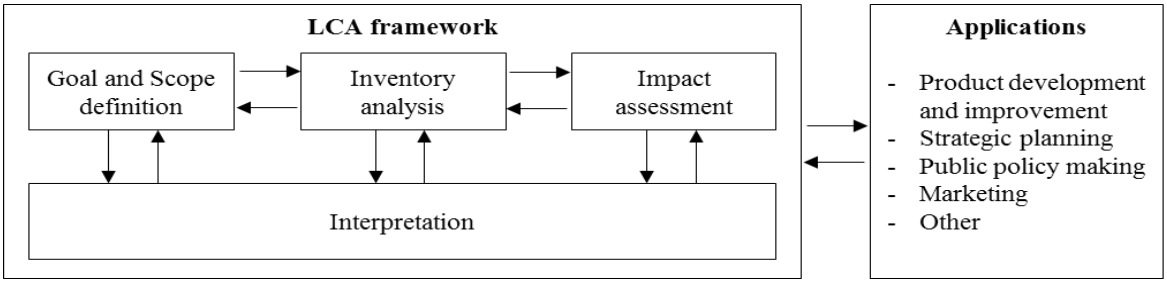


Image adapted from: ISO, 2006

Figure 3.1: Overview of LCA stages and applications

3.2.2 Goal and scope definition

The first step of any LCA is the goal and scope definition. The goal definition describes the LCAs context and purpose as well as how and by whom the results will be used. The scope definition provides further detail on what is covered in the LCA. The individual aspects that the ISO standards (ISO 2006; ISO 2018) and the ILCD handbook (European Commission, 2010) request to be included in the goal and scope definition are summarised in Table 3.1.

Table 3.1: Aspects for the goal and scope definition

Aspect covered			ISO	ILCD
Goal	-	Intended applications / disclosure to the public	X	X
	-	Intended audience	X	X
	-	Reasons for carrying out the study / decision context	X	X
	-	Key limitations / commissioner of the study	-	X
Scope	-	Product system and functional unit	X	X
	-	Modelling framework	-	X
	-	Allocation procedures	X	X
	-	System boundaries	X	X
	-	Data requirements, assumptions and limitations	X	X
	-	Selection of impact categories and methods	X	X
	-	Type of critical review and format of the study	X	X

Intended applications / disclosure to the public

LCAs have many applications including a weak point analysis, comparisons and benchmarking (European Commission, 2010). If public comparative assertions are to be made, it is required to conduct a critical review and to select an adequate set of internationally accepted impact categories for the impact assessment (ISO 2006; ISO 2018). In addition, a sensitivity and uncertainty analysis shall be carried out while a weighting shall not be used (Ibid).

Intended audience

The intended audience can be divided into internal and external as well as technical and non-technical groups (European Commission, 2010). Their early statement is important as it permits to identify confidentiality and review needs.

Reasons for carrying out the study / decision context

Common reasons to carry out LCA studies are accounting purposes or a decision support (European Commission, 2010). It is important to state the reason as it influences the choice of the allocation procedure and modelling framework. For example, a study that supports a decision with small scale market consequences may apply a system expansion and an attributional approach (Figure 3.2, Laurent et al., 2014a).

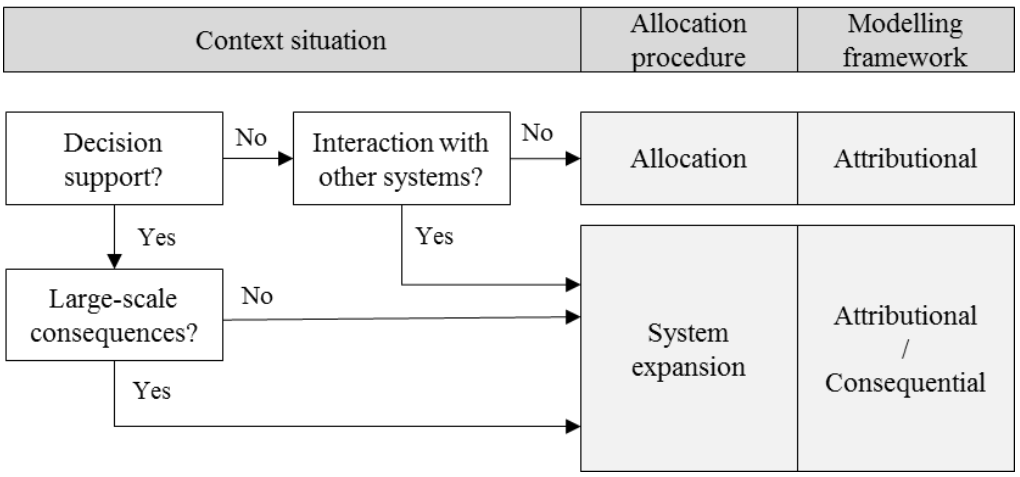


Image adapted from Laurent et al., 2014a

Figure 3.2: Decision context

Key limitations / commissioner of the study

Key limitations include the narrow choice of impact categories such as in carbon footprints or the inappropriate selection of time, location or product data (European Commission, 2010). For example, when an LCA study based on average data from a wider region shall be used to inform decisions on a small island with its own characteristics, this needs to be stated early on. In addition, influential stakeholders that commissioned or co-financed the study shall be named.

Product system and functional unit

A description of the product system and its function provides the context for the analysis. To compare different product systems, it is critical to define a functional unit (ISO, 2006) so that a functional equivalence of the different systems is achieved. For example, if an incineration plant is to be compared with a landfill, a typical functional unit would be the treatment of one tonne of a specific waste (Laurent et al., 2014a). Additional functions like the energy production from the incineration plant can be subtracted from the system.

Modelling framework

For the modelling framework it is distinguished between attributional and consequential LCAs. Attributional LCAs assume a fixed market situation based on historic data whereas in consequential LCAs the effects of changes are included in the model in a dynamic way (European Commission, 2010). Thus, consequential LCAs are particularly suitable for decisions with large-scale consequences like infrastructure projects for a future energy demand, while attributional LCAs are more suitable for accounting purposes or decisions in which the market remains largely unchanged.

Allocation procedures

Allocation procedures assign impacts from processes with multiple in- or outputs. For example, when an incineration plant ("A" in Figure 3.3) that produces steam and electricity ("X" and "Y" in Figure 3.3) is compared with other electricity producing technologies, it needs to be determined how much of its impact ("Z" in Figure 3.3) occurs for the electricity production alone. While allocations can be conducted on a mass or economic basis, it is generally recommended to avoid them (ISO, 2018). Apart from dividing the unit process into sub-processes, this can also be achieved by a system expansion that subtracts an alternative single in- or output process ("B" in Figure 3.3) from the multi-functional process as indicated in Figure 3.3.

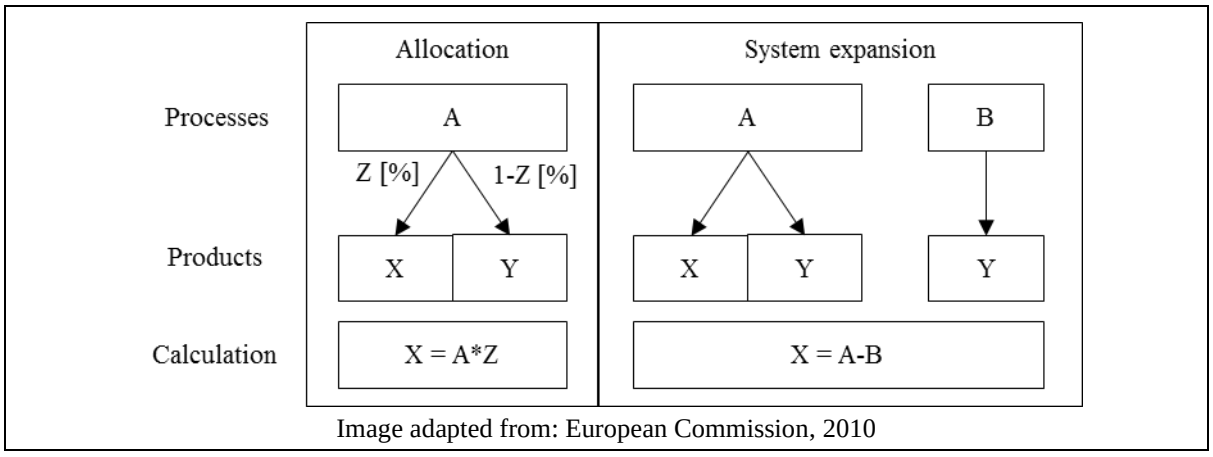


Image adapted from: European Commission, 2010

Figure 3.3: Allocation and system expansion

System boundaries

System boundaries highlight the included processes. They are typically illustrated with help of a process flow diagram that covers the different unit processes and material flows. Process exclusions from the system boundaries need to be clearly stated. Examples for this are secondary transport processes or the production of capital goods such as vehicles, harbours or roads (Laurent et al., 2014a).

Data requirements, assumptions and limitations

The main data types and sources are identified as part of the scope definition to ensure that it meets the studies time, geography and technology related scope (European Commission, 2010).

Selection of impact categories and methods

The selection of impact categories and methods shall be made and documented during the scope definition to avoid a later influence on the results (European Commission, 2010). It is generally recommended to use internationally accepted impact methods and to cover all

relevant impact categories. Impact assessment methods can be divided into mid- and end-point methods. Midpoint methods typically contain a wide range of impact categories such as climate change and human toxicity while endpoint methods focus on areas of protection such as human health, the natural environment and natural resources. This makes midpoint methods more accurate and precise but also less comprehensible than endpoint methods.

Type of critical review

Critical reviews are carried out by independent external reviewers with previous experience and system expertise. For studies that include comparative assertions, relevant stakeholders should also be involved (European Commission, 2010). While the review is carried out at the end of an LCA, the review type is specified as part of the scope (ISO, 2006). Although critical reviews are formally required by the ISO standard (ISO, 2006), only few publications carry out an external peer-review (Laurent et al., 2014a).

3.2.3 Inventory analysis

During the inventory analysis the actual data is collected. This includes process specific raw material and energy inputs as well as effluents and output waste. At this stage foreground processes which fulfil the functional unit; and background processes that supply the required energy and raw materials are distinguished (European Commission, 2010). In general, specific primary data is only collected for the foreground processes, while secondary data is used for the background processes. The data is then transferred into in- and output tables to calculate and aggregate the overall inventory of the product system. While this is possible by hand, LCA software such as GABI or SimaPro is typically used.

A common problem during the inventory analysis is the availability of representative and appropriate data. For example, when data from small scale experiments is used to model large scale technologies, a significant level of uncertainty is introduced. One way to deal with data uncertainties is the assignment of uncertainty ranges. For this a best- and worst-case with minimum and maximum values can be used (Laurent et al., 2014a). Inventory data is commonly divided into short- and long-term emissions depending on whether they occur within or after the first 100 years. As long-term emissions are very uncertain only short-term emissions are used for the impact assessment (European Commission, 2010).

3.2.4 Impact assessment

The impact assessment consists of several steps that are depicted in Figure 3.4.

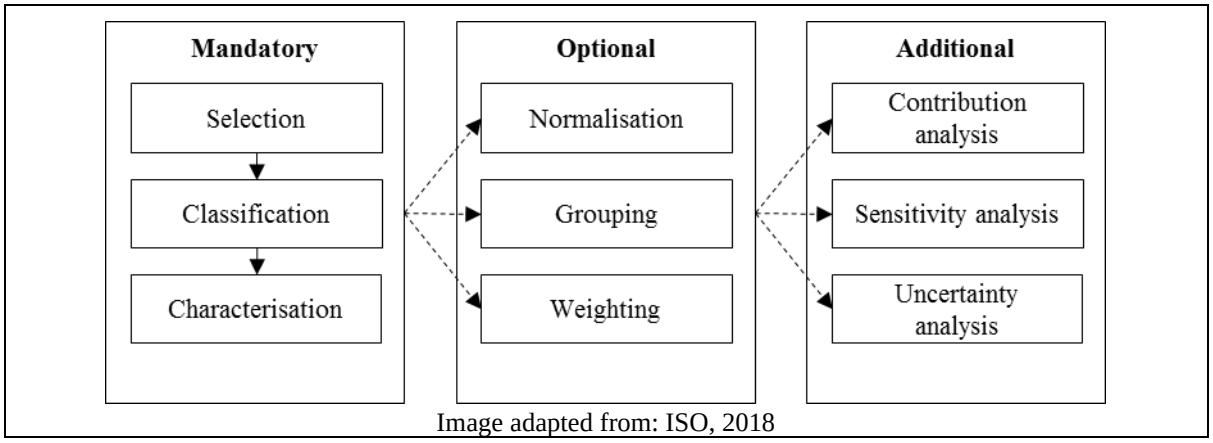


Image adapted from: ISO, 2018

Figure 3.4: Steps of the life cycle impact assessment

Mandatory steps

The mandatory steps include the selection of impact categories and methods (see goal and scope definition above) as well as a classification and characterisation (ISO, 2018). The classification entails the assignment of inventory results to the selected impact categories. During the characterisation the inventory results are multiplied with characterisation factors. For example, methane may be multiplied with a characterisation factor of 25 kg CO_2-eq/kg to obtain its equivalent impact on climate change (European Commission, 2010). For each impact category the results are then aggregated and presented.

Optional steps

Normalisation, grouping and weighting are optional steps that aim to facilitate a decision making (ISO, 2018). Grouping entails the sorting or ranking of impact categories while a normalisation involves the comparison to a reference case. A typical reference case would be the impacts from an average European citizen over one year. This comparison then allows to identify impact areas with great differences to that norm. Possibly the most debated approach is weighting, were impact categories are given weights depending on their relative importance. As weighting is very subjective, it shall not be used in LCAs that make public comparative assertions (Ibid).

Additional steps

Additional impact assessment steps are a contribution, sensitivity and uncertainty analysis which provide the basis for the interpretation phase (ISO, 2018). A contribution analysis identifies the highest contributions in each impact category while an uncertainty and sensitivity analysis show how uncertainties and changes affect the results.

3.2.5 Interpretation

The last phase of an LCA is called interpretation. As part of this the inventory and impact assessment results are evaluated against the goal and scope definition. The interpretation covers the (1) identification of the most important issues; an (2) appraisal of the completeness, consistency and robustness; and the (3) forming of conclusions and recommendations (ISO, 2018).

3.3 Relevant LCAs for this thesis

3.3.1 LCAs for marine litter management

To identify LCAs for the management of marine litter, a literature search was conducted on Scopus and Google scholar using the title keywords "life cycle assessment", "LCA", "whole system" or "Systems thinking" in combination with "marine litter", "marine debris" or "plastic debris". Until January 2020 two relevant studies came out of this search. The first study, calls on stakeholders to develop new methodologies for the integration of marine litter into LCA (Sonnemann and Valdivia, 2017) while the second study proposes a life cycle impact assessment model that links plastic waste entanglements to a biodiversity loss on a geographical and taxa specific scale (Woods, Rødder and Verones, 2019). Although both studies are relevant for the methodological development of LCA, they do not apply LCA.

3.3.2 LCAs for end-of-life fishing gear management

The European end-of-life fishing gear (EOLFG) market is dominated by three companies namely Nofir, Aquafil and Plastix (Chapter 2.1.4). To find LCA studies on the waste management of EOLFG an internet search on Google was conducted in December 2019, using the key words LCA and the corresponding company names. In all three cases information about an LCA study could be obtained.

1. The study from Nofir uses 1kg of average output products in form of nylon (76.2%), polypropylene (12.6%), polyethylene (8.7%), lead (1.9%) and steel (0.6%) as functional unit to compare two waste treatment scenarios (Nofir, 2015). The first scenario represents Nofir's actual process during which 76% of EOLFG are recycled; 22% disposed and 2% reused; while the second scenario assumes a default situation during which 20% are recycled, 45% disposed and 35% dumped at sea. Focusing on energy consumption, the study concludes that scenario 1 and 2 lead to a carbon footprint of 1.89 kg CO_2-eq/kg and 5.5 kg CO_2-eq /kg respectively.

2. Aquafil conducted an LCA to compare their production of Econyl yarn with a conventional nylon production. They found that their processes reduce the global warming potential by up to 80% (Aquafil, n.d). They also published an environmental product declaration that provides further detail on their process. Interestingly, the production of 1 kg of Econyl yarn requires 1.72 kg of nylon waste, 0.869 MJ of direct energy and 8.76 kg of direct water (Aquafil, 2018c). The activities at the recycling plant dominate the potential environmental impact with a contribution of 1.53 kg CO_2-eq/kg to the global warming potential, 4.4 g SO_2-eq/kg to the acidification potential and 1.7 g P-eq/kg to the eutrophication potential.

3. The LCA study on Plastix uses the functional unit of 1 tonne output material to compare the production of virgin and recycled material for nylon, high-density polyethylene and polypropylene each (Storm, 2017). The study finds that the energy consumption for the recycled plastics lies between 0.922 and 1.26 kWh/t while the energy consumption for virgin polymers ranges between 8.75 and 25.1 kWh/t. A sensitivity analysis with increased transport distances and a higher contamination in form of sand is conducted without affecting the overall results.

While all three studies highlight the benefit of recycling in comparison to a virgin polymer production or a default disposal scenario, they do not present a detailed life cycle inventory so that a comparison with other studies is difficult to make.

3.3.3 LCAs for solid waste management

LCA studies in the area of solid waste management are numerous and thus frequently reviewed. Table 3.2 provides an overview of review studies on Scopus that matched the title key words "LCA" or "life cycle assessment" in combination with "solid waste review".

Table 3.2: LCA review studies on solid waste

Authors, year	Year	Focus	Studies reviewed	Period covered	Citations
Khandelwal et al.	2019	Municipal solid waste	153	2013-2018	27
Li et al.	2019	Waste for high-way pavement	34	2006-2016	10
Yadav and Samadder	2018	Solid waste in Asia	91	2006-2017	24
Yadav and Samadder	2017	Municipal solid waste	30	2003-2014	12
Laurent et al.	2014	Application	222	1995-2012	248
Othman et al.	2013	Solid waste in Asia	7	2006-2009	76

The most extensive reviews were carried out by Laurent et al. (2014a; 2014b) and Khandelwal et al. (2019) covering 375 studies from complementing time frames (Table 3.2). As this makes them the most representative, some of their key findings are further introduced. The studies:

- gradually increased over time (from two publications in 1995 to over 20 after 2009),
- covered Europe (54%), Asia (31%), North America (7%) and other regions (8%),
- mainly evaluated mixed, plastic or organic waste, and
- focused on incineration (62%), landfilling (59%) and mechanical recycling (47%).

Within the reviewed studies the ISO standard and ILCD guidelines were not always followed. According to Laurent et al. (2014a) and Khandelwal et al. (2019), some of the most common methodological shortcomings included:

- a missing goal definition and unclear system boundaries,
- unrepresentative inventory data,
- insufficient impact coverage, and
- a lack of a sensitivity or uncertainty analysis.

Thus, Laurent et al. (2014b) classified 135 studies as poor and neglected them for further analysis. From the remaining mid- and good quality studies 34 made comparisons generally supporting the waste hierarchy for plastics, paper and mixed waste (Laurent et al., 2014b). However, an overview of the mid- and good quality studies was not provided though.

Plastic waste

To identify relevant LCA studies on plastic waste, a literature search for articles on Scopus using the title key words "LCA" or "life cycle assessment" in conjunction with "plastic* waste" was conducted in December 2019. To focus on a manageable amount of relevant studies, publications before 2013 or with a geographical scope outside Europe were excluded, while ISO-compliant studies published before 2013 (Laurent et al., 2014a) were included when otherwise meeting the search scope. This led to 22 studies which were critically reviewed (Table 3.3).

All reviewed studies made public comparisons between integrated waste management systems or individual technologies (Table 3.3). The functional units were clearly stated, typically in form of one tonne of a specific waste stream (Table 3.3). All studies applied system expansion and provided a clear process flow diagram for the system boundaries (Table 3.3).

Table 3.3: Summary of review findings for selected LCA studies

References		1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18	19	20	21	22
Product System	Reuse / Prevent	-	-	-	-	-	-	-	-	-	-	-	-	X	-	-	X	-	-	-	-	-	-
	Composting	-	-	X	(X)	-	X	-	-	(X)	-	-	-	-	X	X	-	-	(X)	-	-	-	-
	Mech. Recycling	X	X	X	(X)	-	-	X	-	(X)	X	(X)	X	(X)	X	-	(X)	(X)	X	X	(X)	(X)	X
	Chem. Recycling	X	-	-	-	X	-	-	-	-	-	-	-	-	-	-	-	(X)	-	-	(X)	(X)	X
	Energy Recovery	(X)	(X)	X	(X)	X	X	(X)	X	(X)	X	(X)	X	(X)	X	X	(X)	(X)	X	(X)	X	(X)	X
	Disposal	(X)	(X)	X	(X)	X		(X)		(X)	-	(X)	(X)	(X)	X	X	(X)	(X)	X	(X)	X	(X)	X
FU	Waste material	P+	P+	P+	P+	P	P+	P+	P+	P+	P+	P+	P+	P+	P+	P+	P	P+	P+	P+	P	P+	P+
	Quantity	1t	1t	1t	1t	1t	O	1t	O	O	1t	1t	1t	O	O	1t	O	O	O	1t	1t	O	1t
	Framework	C	-	A	-	-	-	A	A	-	A	-	C	-	-	A	-	A	A	C	-	-	-
	Allocation	SE	SE	SE	SE	SE	SE	SE	SE	SE	SE	SE	SE	SE	SE	SE	SE	SE	SE	SE	SE	SE	SE
Data	Process graph	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X
	Primary data	-	-	X	X	-	-	X	-	-	-	X	-	-	-	-	-	X	X	-	X	X	X
	Clear inventory	X	-	-	-	-	-	-	Pa	-	Pa	Pa	Pa	-	-	X	-	X	X	Pa	X	X	X
Impact categories	Software	Easetech	SimaPro	Gabi	Gabi	Gabi	-	Gabi	SimaPro	WRATE	-	SimaPro	STAN	SimaPro	-	SimaPro	SimaPro	Gabi	SimaPro	Easetech	Gabi	Gemis	Gabi
	Method	ILCD	CML	ReCiPe	CML	CML	PEF	ReCiPe	ReCiPe	-	ReCiPe	PEF	-	ILCD	Impact	CML	ILCD	CML	ILCD	EDIP	CML	-	CML
	Global warming	X	X	X	X	X	X	X	X	X	X	X	-	X	X	X	X	X	X	X	X	X	X
	Acidification	X	X	X	X	X	X	-	X	X	X	X	-	X	X	X	X	X	X	X	X	X	X
	Eutrophication	X	X	X	X	X	X	-	X	-	X	X	-	X	X	X	X	X	X	X	X	-	-
	Ozone depletion	-	X	X	-	X	X	-	-	-	X	X	-	X	X	X	X	-	-	-	X	-	-
	Photo-chemical oxidant formation	X	X	X	X	X	X	-	X	-	X	X	-	X	-	X	X	X	X	X	X	X	-
	Particulate matter	-	-	-	-	-	X	-	-	-	-	X	-	X	-	-	X	-	-	-	-	-	-
	Human toxicity	X	X	X	X	X	X	-	X	-	X	X	-	X	X	-	X	-	X	X	X	-	-
	Ecotoxicity	-	X	X	-	X	X	-	X	-	X	X	-	X	X	-	X	-	X	X	-	-	-
	Fossil depletion	-	X	X	-	X	X	-	X	X	X	X	-	X	X	X	X	-	X	-	X	-	-
	Metal depletion	-	X	-	-	X	X	-	X	X	-	X	-	X	X	X	X	-	X	-	X	-	-
	Water depletion	-	-	-	-	-	X	-	X	-	-	X	-	X	X	-	X	-	-	-	-	-	-
	Other	-	-	-	-	-	X	-	-	-	-	X	X	X	X	X	X	-	X	-	X	X	X
Contribution analysis		X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	-	X	X	X	X	X	X
Sensitivity analysis		X	-	X	-	-	X	X	X	X	X	X	X	-	X	X	X	X	-	X	X	X	X
Uncertainty analysis		X	-	-	-	-	-	-	-	-	-	-	X	-	-	-	-	-	-	-	-	-	-
Completeness grade		1	0	0	0	0	0	0	1	0	1	1	0	0	0	1	0	1	0	1	1	1	1

[1] Faraca, Martinez-Sanchez and Astrup, 2019; [2] Fiore et al., 2019; [3] Beigbeder et al., 2019; [4] Cremiato et al., 2018; [5] Gear et al., 2018; [6] Fieschi and Pretato, 2018; [7] Unger et al., 2017; [8] Ripa et al., 2017; [9] Tunesi, Baroni and Boarini, 2016; [10] Wäger and Hischier, 2015; [11] Biganzoli et al., 2015; [12] Laner et al., 2015; [13] Nessi, Rigamonti and Grosso, 2015; [14] Rossi et al., 2015; [15] Quirós et al., 2015; [16] Nessi, Rigamonti and Grosso, 2014; [17] Al-Salem, Evangelisti and Lettieri, 2014; [18] Ferreira et al., 2014; [19] Rigamonti et al., 2014; [20] Shonfield, 2008; [21] Jenseit et al., 2003; [22] Kreißig et al., 2003;

"x": covered; "(X)" covered in system; "-" not covered; "P": Plastics; "P+": Plastics and other; "C": Consequential; "A": Attributional; "SE": System expansion; "Pa": partially; "O": Other

A contribution, sensitivity and uncertainty analysis were missing in 5%, 23% and 91% of the studies respectively (Table 3.3), while primary data, a clear inventory and an analysis of more than one impact category were lacking in 59%, 45% and 9% of the studies respectively (Table 3.3). Only ten studies specified the modelling framework from which seven used an attributional and three a consequential approach (Table 3.3). The ten studies that provided inventory data, a sensitivity analysis and a sufficient impact coverage were classified as complete (Table 3.3).

As suggested by Laurent et al. (2014a) the results of the complete studies were further reviewed. Although the waste hierarchy was generally supported, a few exceptions in specific impact categories were found. Those include a:

- lower global warming potential for landfills compared to incineration (Shonfield, 2008),
- lower ozone depletion potential for incineration compared to a mechanical recycling (Wäger and Hischier, 2015),
- lower photochemical oxidant formation, human toxicity, eutrophication and ozone depletion potential for a chemical recycling compared to a mechanical recycling (Faraca, Martinez-Sanchez and Astrup, 2019; Quirós et al., 2015).

This indicates that a decision based on the potential environmental impacts may not always be straightforward to make. Each LCA is case specific and general outcomes should not be assumed.

3.4 Chapter summary

This chapter introduced LCA as key methodological tool to assess the environmental performance of systems. The origins of LCA and its standard methodology were briefly introduced. Although applied LCA studies on marine litter could not be detected, LCA studies on end-of-life fishing gear and plastic waste were identified and reviewed. This revealed a general lack of transparent and high-quality inventory data which is needed for a meaningful LCA. To identify suitable waste treatment options for DFG, new data needs to be collected, analysed and communicated in a comprehensive and reproducible way.

4 Methodology

4.1 General setup of experiments and LCA

To realise the research objectives (Chapter 1.3) a combination of industrial experiments and LCA was used. The industrial experiments were initiated by WWF Germany and PreZero with the aim to evaluate the technical feasibility of recycling options for DFG. This mainly included a pre-treatment, mechanical recycling and gasification. The author of this thesis was brought in to collect LCA relevant data during the experiments and to subsequently compare the waste treatment techniques. While a potentially harmful incineration or landfilling experiment was not conducted, those waste management options were modelled theoretically to cover the most common waste treatment techniques.

The methodological setups of the industrial experiments are described in the respective sections of Chapter 5-7. The main purpose of the experiments for this thesis was to establish a typical DFG waste composition and to collect process data. As some experimental processes were carried out manually, scale-up considerations for a fair comparison with established technologies needed to be made. Based on this information the LCA model is built.

The LCA generally follows the guidelines set out in ISO 14040 (ISO, 2006), ISO 14044 (ISO, 2018) and the ILCD handbook (European Commission, 2010). As such it covers four parts namely the (1) goal and scope definition, the (2) inventory analysis, the (3) impact assessment and the (4) interpretation. While this chapter further describes the methodological choices of each LCA phase, the actual inventory analysis is conducted in Chapter 5-9, the impact assessment in Chapter 10 and the interpretation in Chapter 11-12.

Throughout this thesis a uniform numbering format was used. It considers the first three significant figures that would also be used in an exponential notation. For example, the number 123.4 would be expressed as 123, whereas the number 1.234 would be expressed as 1.23. However, as there is much uncertainty in the data, the reader should not mistake the numbering format for an indication of precision. The data uncertainty is assessed in Chapter 10.

4.2 Goal and Scope definition

4.2.1 Modelling framework and allocation procedure

The aim of the LCA study presented in this thesis is to compare alternative waste treatment options for DFG in the EU to help stakeholders decide on suitable waste treatment infrastructure in the future (Chapter 1.3). Due to the small quantities of retrieved DFG (Chapter 2.2.2), such infrastructure is not expected to affect the current energy production or other relevant background systems at a large scale. Thus, it is suitable to conduct an attributional LCA using system expansion (European Commission, 2010).

4.2.2 Product systems and functional unit

As part of this thesis four distinct waste treatment scenarios are compared. Those include a (1) mechanical recycling to produce nylon, a chemical recycling to produce syngas, an (3) energy recovery to produce heat and electricity and a (4) disposal to store DFG.

The functional unit is defined as the treatment of 1000 kg of DFG with an average composition of 1.7% organics, 7.4% steel, 6.8% lead, 41.9% minerals and 42.2% polymers (Table 4.1). The waste composition was established based on the experimental output quantities. The calculation of the waste composition has been included at the end of this thesis (Appendix A).

Table 4.1: Waste composition for trawl and gillnets

Material	Unit	**Organics** (Live and dead fish)	Metals		**Minerals** (Sediments, Stones, Shells)	Polymers		Total
			Ferrous (**Steel**)	Non-ferrous (**Lead**)		Density < 1 (**PE**/**PP**)	Density >1 (**PA**)	
Trawl nets	%	0	7.4	0	56.3	0	36.2	100
Gillnets	%	3.4	7.3	13.5	27.5	4.6	43.6	100
Average	%	1.7	7.4	6.8	41.9	2.3	39.9	100

4.2.3 System boundaries

The system boundaries in Figure 4.1 depict all foreground processes from the retrieval to a disposal or product / energy substitution. For this, it is distinguished between primary, secondary, avoided and excluded processes. Primary processes fulfil the primary goal of the waste treatment scenario and have “DFG” as an input. For scenario 1 the primary processes include the retrieval, sorting, shredding, 1. density separation, 2. density separation, washing, drying and extrusion (Figure 4.1). For scenario 2 the primary processes are the retrieval, sorting and shredding as well as the gasification. In scenario 3 the primary processes entail the retrieval, sorting, shredding, 1. density separation and incineration while scenario 4 involves the retrieval and landfilling (Figure 4.1).

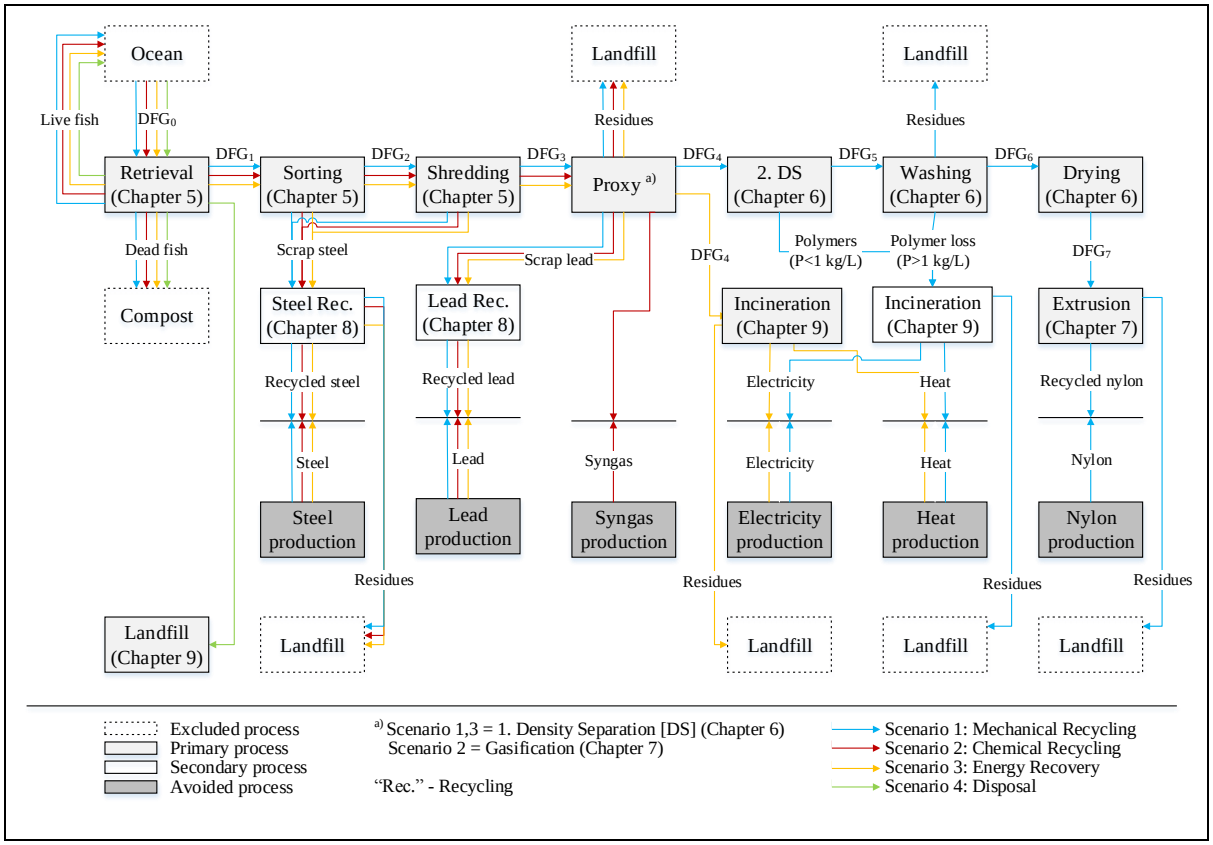


Figure 4.1: System boundaries for the LCA study

Secondary processes are process that treat separated or residual fractions. A steel and lead recycling take place in scenario 1-3 while a secondary incineration is conducted in scenario 1. The landfilling of residues is excluded, because they mainly comprise of inert material for which a low environmental impact over a 100-year period would be expected (i.e. Birgisdóttir et al., 2007). Also, appropriate datasets for example from Ecoinvent could not be found to model this process. Avoided processes are included to solve the multifunctionality of the waste treatment scenarios through a system expansion. Those processes include an average steel, lead and syngas production as well as an average heat, electricity and nylon production.

For a comparison of product systems, processes that are the same in all scenarios can be excluded. While this was done for the release of live fish and the composting of dead fish, it was not done for the retrieval, because of its expected high impact. Capital goods such as infrastructure or machines have not been included, despite their potentially large impact to transport processes (Brogaard and Christensen, 2012). However, this is common practice for LCA studies in the area of waste management (Laurent et al. 2014a), because capital goods do not typically contribute to waste treatment processes at a significant scale (Brogaard, Riber and Christensen, 2013; Brogaard et al., 2013). The transport between the modelled primary and secondary processes was included.

4.3 Life cycle inventory analysis

4.3.1 Data sources and representativeness

For the modelling of primary and secondary processes a combination of experimental, company and literature data were used. More specifically, experimental data was mainly collected to quantify the illustrated material flow (Figure 4.1), to obtain process specific energy consumptions or to model the fate of chemical elements throughout processes. Company and literature data were used to fill gaps for ancillary materials and process emissions. Avoided production processes like the steel production and background processes like the electricity generation were modelled based on the Ecoinvent database.

While all data is provided in Chapter 5-9, key data sources are introduced here. Apart from experimental data, those include the Best Available Techniques reference documents [BREFs], the European Monitoring and Evaluation Programme [EMEP] /European Environment Agency [EEA] air pollutant emission inventory guidebook as well as other literature sources (Table 4.2). In general, the data represents current large-scale production facilities in Europe. When such data was not available, data for example from older sources (i.e. DEFRA, 2004) was used.

The selection of processes from the Ecoinvent 3.5 database followed a similar approach. Within the default attributional system model, preference was given to current market processes representing the average situation in Europe. When those were not available, the global average was chosen. A full list of the selected unit processes from Ecoinvent is provided at the end of this thesis (Appendix B). For the energy production the German electricity mix (Table 10.18) was modelled to represent the location where the DFG was collected and processed.

Table 4.2: Overview of key data sources

Data type	References	Relevant process	Geography	Technology	Time
Experimental	Primary data (including from project partners)	All	Germany, Switzerland	DFG specific	2016-2019
Literature	BREF (2006)	Incineration	Europe	Average	<2006
	BREF (2013)	Steel recycling			<2013
	BREF (2017)	Lead recycling			<2017
	EMEP/EEA (2016a)	Retrieval			<2016
	EMEP/EEA (2016b)	Transport			<2016
	EMEP/EEA (2017)	Incineration			<2017
	Villanueva-Rey et al. (2017)	Retrieval	Spain	Average	<2017
	DEFRA (2004)	Landfill	United Kingdom	Average	<2004

4.3.2 Software choice

To facilitate the collection and analysis of data the PhD version 8.3.0.0 of the LCA software SimaPro was used. While specific software for the modelling of waste treatment scenarios exist,

the choice of software does not influence the results (Laurent et al., 2014a). However, due to its detailed uncertainty analysis capability, SimaPro was used for this thesis.

4.3.3 Uncertainty and assumptions

To address quantitative uncertainties of the input values, ranges are assigned to the inventory data of the foreground system defining their probability distribution (Bisnella et al., 2016). In this thesis, the uncertainty was modelled based on a triangular distribution with min, max and average values. When those values could not be obtained, probability distributions were assigned following the pedigree approach (Weidema et al., 2012). As part of this, the data's reliability and completeness as well as its temporal, geographical and technological representativeness are evaluated assigning scores from one to five (see Table 4.3). In addition, a basic uncertainty value is allocated in reference to the data type. This and the scoring results are then transferred into a log-normal probability distribution. More detailed information on the approach can be found in Weidema et al. (2012).

Table 4.3: Pedigree matrix scores to assess the data quality

Score	1	2	3	4	5
Reliability	Verified data based on measurements	Verified data partly based on assumptions or non-verified data based on measurements	Non-verified data partly based on qualified estimates	Qualified estimate (e.g. by industrial ex-pert)	Non-qualified estimate
Completeness	Representative data from all sites relevant for the market considered, over an adequate period to even out normal fluctuations	Representative data from >50% of the sites relevant for the market considered, over an adequate period to even out normal fluctuations	Representative data from only some sites (<<50%) relevant for the market considered or >50% of sites but from shorter periods	Representative data from only one site relevant for the market considered or some sites but from shorter periods	Representativeness unknown or data from a small number of sites and from shorter periods
Temporal correlation	Less than 3 years of difference to the time period of the dataset	Less than 6 years of difference to the time period of the dataset	Less than 10 years of difference to the time period of the dataset	Less than 15 years of difference to the time period of the dataset	Age of data unknown or more than 15 years of difference to the time period of the dataset
Geographical correlation	Data from area under study	Average data from larger area in which the area under study is included	Data from area with similar production conditions	Data from area with slightly similar production conditions	Data from unknown or distinctly different area
Technology correlation	Data from enterprises, processes and materials under study	Data from processes and materials under study (i.e. identical technology) but from different enterprises	Data from processes and materials under study but from different technology	Data on related pro-cesses or materials	Data on laboratory scale or different technology

Table adapted from: Weidema et al., 2012

4.3.4 Scale-up

The collected data for the recycling pathways in this thesis mainly derive from small-scale industrial processes. However, such initial processes are not yet designed to process waste in the most efficient way. To allow a comparison with more developed disposal or energy recovery techniques, the small-scale processes needed to be scaled-up. While the waste composition specific data remained unchanged, the energy and ancillary product consumption as well as the emission data of small-scale processes is adequately adapted throughout the research.

4.4 Life cycle impact assessment

4.4.1 Impact methodology

The choice of impact methodology can have large influence on the results (i.e. Dreyer, Niemann and Hausschild, 2003) and should thus be considered carefully. Traditionally, the most common impact methods in the area of solid waste were CML (31%), EDIP (21%) and Eco-indicator (14%) (Laurent et al., 2014a). In 2008, the CML and Eco-indicator methods were merged into a new impact methodology called ReCiPe (Khandelwal et al., 2019). In 2011, the European Commission compared various impact methods and published recommendations for their use. While ReCiPe was not the first choice for all impact categories, it was classified as very robust across a wide range of impact categories (European Commission, 2011). Due to its robustness and increasing uptake in the literature (Khandelwal et al., 2019), ReCiPe is chosen as impact method for this thesis. For the modelling ReCiPe 2008 Version 1.12 is used.

Approach to uncertainty

Like all impact methods, the characterisation model of ReCiPe is subject to uncertainty (i.e from an incomplete knowledge of the environmental mechanisms). To address this uncertainty, it is distinguished between an individualist, hierarchist and egalitarian approach (Goedekamp et al., 2013). The individualist approach adopts a short-term (20 year) perspective using certain scientific knowledge while being optimistic about technological change. The hierarchist approach adopts a 100-year time frame acknowledging the political directions. The egalitarian approach adopts the most protective view as it considers 500 years. In line with the 100-year time frame in other impact methods such as from the Intergovernmental Panel on Climate Change (IPCC), this thesis adopts a hierarchist approach.

4.4.2 Impact categories

ReCiPe covers 18 impact categories at midpoint level (Table 4.4) and links them to three areas of protection at an endpoint level, namely (1) human health, (2) ecosystem diversity and (3) resource availability (Goedkoop et al., 2013).

Table 4.4: List of ReCiPe impact categories

1	Climate change	7	Terrestrial ecotoxicity	13	Ozone depletion
2	Terrestrial acidification	8	Freshwater ecotoxicity	14	Particulate matter formation
3	Freshwater eutrophication	9	Marine ecotoxicity	15	Ionising radiation
4	Marine eutrophication	10	Water depletion	16	Agricultural land occupation
5	Photochemical oxidant formation	11	Metal depletion	17	Urban land occupation
6	Human toxicity	12	Fossil depletion	18	Natural land transformation

In this thesis only impact categories at midpoint level are considered as they allow a more detailed evaluation. From those, agricultural land occupation, urban land occupation and natural land transformation were excluded due to their weak methodologies (European Commission, 2011). In addition, ozone depletion, particulate matter formation and ionising radiation were neglected as well. This is because those impact categories are considered the least relevant (Zampori et al., 2016) being rarely covered in scientific studies (Laurent et al., 2014a; Khandelwal et al., 2019; Table 3.3). For example, ozone depletion has already been addressed by international legislation and possible impacts in this category would likely not be representative for the situation today. The remaining impact categories are briefly introduced.

Climate change

Climate change, caused by greenhouse gas emissions and a consequent global warming, affects both human health and ecosystem diversity for example through increasing droughts, flooding or wildfires (Goedkoop et al., 2013). To characterise the climate change impacts at a midpoint level in ReCiPe, the widely accepted global warming potential [GWP] for greenhouse gasses in kg CO_2-eq is used.

Terrestrial Acidification

Terrestrial acidification describes the deposition of atmospheric substances such as nitrates, sulphates and phosphates in the terrestrial environment leading to an increase of soil acidity and a potential decline of plant species (Goedkoop et al., 2013). Expressed in kg SO_2-eq, its midpoint characterisation factors in ReCiPe are calculated based on a combination of fate, deposition and soil saturation models.

Eutrophication

Eutrophication stems from an increasing supply of nutrients to water bodies that can lead to algal blooms and a subsequent oxygen depletion (Goedkoop et al., 2013). While phosphorous controls biomass growth in freshwater, nitrogen is mainly responsible for eutrophication in marine waters. Consequently, ReCiPe characterises freshwater and marine eutrophication based on fate models in kg P-eq and kg N-eq respectively.

Photochemical oxidant formation

Photochemical oxidants like ozone are formed in reactions of NOx and non-methane volatile organic compounds [NMVOC] and they have the potential to cause human health damages such as lung inflammations (Goedkoop et al., 2013). The midpoint characterisation factors in ReCiPe are calculated based on the relative formation of ozone from individual substances expressed as kg NMVOC-eq.

Toxicity

Toxicity describes the degree of a substance's toxic quality to harm organisms. In ReCiPe it is distinguished between human toxicity and freshwater, marine and terrestrial ecotoxicity (Goedkoop et al., 2013). Based on fate and effect models, toxicity is expressed as kg 1,4-di-chlorobenzene [DB]-eq for different substances at midpoint level.

Water depletion

Water scarcity is an increasingly common phenomena that poses a threat to human health and ecosystems. At midpoint level, ReCiPe provides information about the total use of water in m^3 (Goedkoop et al., 2013).

Metal depletion

Metals are extracted from minerals involving a mining operation. As mineral resources are limited, the cost of mining increases with growing scarcity. Thus, ReCiPe adopts a characterisation factor based on the relative cost increase from mining of a specific element (Goedkoop et al., 2013). In reference to iron, the midpoint characterisation factor is expressed as kg Fe-eq.

Fossil depletion

Fossil fuels are hydrocarbon rich resources such as coal. Its midpoint characterisation factors are based on the lower heating value an expressed in kg oil-eq (Goedkoop et al., 2013).

4.4.3 Impact assessment steps

The mandatory classification and characterisation step were carried out through SimaPro and its results were added to the end of this thesis (Appendix C). Based on those results a contribution analysis is conducted in Chapter 10.1 to compare the waste treatment options and to highlight significant impact contributions. To investigate the robustness of key input parameters a sensitivity analysis is carried out in Chapter 10.3. As variable input parameters the (1) waste composition, (2) technology and market assumptions, (3) transport distances and the (4) underlying energy mix were selected. Ultimately, an uncertainty analysis is conducted in Chapter 10.4 to identify areas of high uncertainty. In line with previous studies (i.e. Faraca, Martinez-Sanchez and Astrup, 2019), this was done as a Monte Carlo simulation with 10,000 sampling points for each waste treatment scenario (Chapter 10.4).

4.5 Interpretation

The interpretation phase identifies the most critical issues, evaluates the completeness, sensitivity and robustness of the results while also providing conclusions and recommendations. In this thesis, the interpretation phase spreads across the discussion and conclusion in Chapter 11 and 12.

4.6 Chapter summary

This chapter provided a general overview of the methodical choices and procedures used to accomplish the research objectives. The goal of the LCA study – to compare alternative waste treatment options for DFG in Europe – and its scope were clearly defined. In addition, the most critical data collection and modelling choices were explained. The impact methodology ReCiPe and its twelve most robust and accepted impact categories were selected and described for the analysis while further information on the impact assessment and interpretation phase were given.

Having identified the need for the establishment and thorough assessment of a DFG waste management system in Europe (Chapter 2-3), this chapter provided the methodological overview to address this. The following chapters will outline the detailed analysis before the results are given, discussed and transferred into conclusions at the end of this thesis.

5 Retrieval, Sorting and Shredding

5.1 Retrieval

5.1.1 Experiment

Context and aim

As part of MARELITT Baltic, WWF Germany conducted two retrievals in 2016 which provided the material for the waste treatment experiments described throughout this thesis. They took place near the coast in high density fishing areas - the first close to Sassnitz and the second near Ahlbeck. They are described in detail to determine the relevant in- and output flows for the life cycle inventory.

Materials and setup

For the retrieval an 8m long aluminium diving vessel and 2-3 scientific divers were deployed (Stolte, 2019a). In addition, an 18m long metal fishing vessel carrying 2-3 fishers was used. The fishing vessel was equipped with a winch and the retrieval gear. The retrieval gear comprised of a hooked steel bar and an attached rope.

Process description

Prior to the retrieval, WWF Germany received information about the exact location of the DFG. Near Sassnitz, the DFG was spotted by divers on the seafloor (Stolte, 2019a) at approximately 16m depth (Stolte, 2019b). Near Ahlbeck, the German Fisheries Control Authority reported multiple DFG locations in a depth of 3-5m (Ibid).

During the planning of the retrieval no legal, environmental or safety restrictions were noted. Therefore, both DFG sites were deemed suitable for retrieval operations.

For the retrieval a combination of diving and bottom trawling was used. After arriving at the indicated site (Figure 5.1), the diving crew verified the DFG location, freed it from obstacles and marked it with buoys (Stolte, 2019a). Then, the fishing vessel released the retrieval gear into the water and navigated through the buoys. After the retrieval gear hooked onto the DFG it was lifted on board with help of the fishing vessel's winch. The time of the process was frequently recorded (Stolte, 2019b; Table 5.1).

On the way back to the harbour, the DFG was roughly sorted on board. If present, live fish were cut free and returned to the ocean, whereas dead fish were removed in various stages of

decomposition and disposed at the harbour. At the harbour, the DFG was cut into smaller pieces, stored inside big bags and weighted.

Technical challenges and process data

Near Sassnitz each retrieval took between 2-3 hours and near Ahlbeck, one retrieval was recorded at 5 hours (Table 5.1). On average, 2-4 dragging attempts were needed before the retrieval gear successfully hooked onto the DFG (Stolte, 2019a).

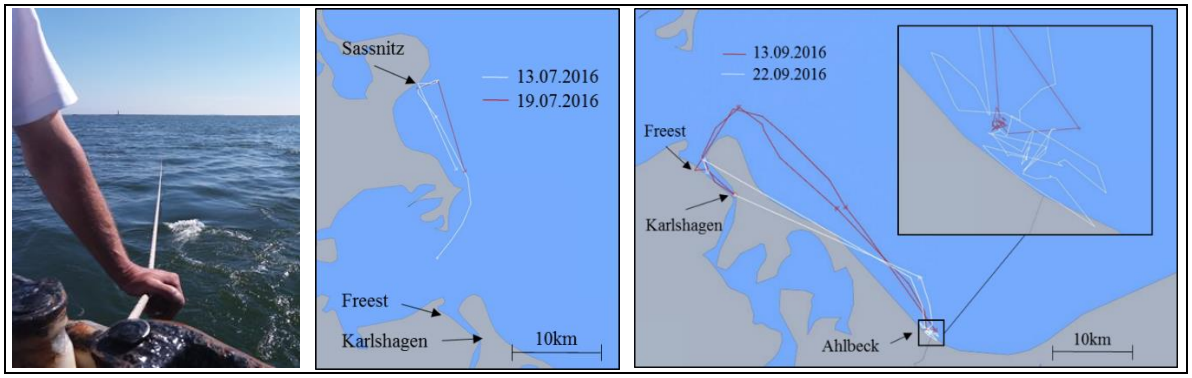


Figure 5.1: Location and travel route for the retrieval operations

Table 5.1: Recorded retrieval time

Location near:	Sassnitz		Ahlbeck	
Date:	13.07.2016	19.07.2016	13.09.2016	22.09.2016
Start of dragging:	08:00	10:00	11:00	-
End of dragging:	10:55	12:00	16:00	-
Duration:	2:55	2:00	5:00	-

Table adapted from: Stolte, 2019b

Output analysis

Near Sassnitz, 2.96 tonnes of DFG were retrieved (Stolte, 2019a). They mainly contained tangled trawl nets, but also metals in form of anchors and chains, rubber coated aluminium and copper cables. A firehose and mussels were also discovered but there were no fish.

The retrieved DFG from Ahlbeck had a total weight of approximately 1.75 tonnes (Stolte, 2019a). It mainly comprised of monofilament gillnets, wooden sticks and fish. The amount of live and dead fish was estimated based on its volume as 20 and 40 kg respectively (Ibid).

The water content in the DFG output was not evaluated. Still, typical water contents of shredded DFG were determined as 1.9-3.5% in a semi-dry and as 18.9-25.0% in a wet condition (Table 5.11). Given that the initially wet DFG had time to dry, the reported DFG fractions are assumed to represent a water content of 3%.

Waste composition

The reported "total" and "fish" weight is summarised in Table 5.2. The difference between the two fractions represents the DFG output weight including big bags. To determine the DFG composition, the weight of the big bags was excluded. Based on the sorting experiments (Chapter 5.2; Table 5.6), it was assumed that a single big bag weighs 1 kg and that approximately 40 big bags were needed to collect 5t of DFG. To account for materials that dropped on the floor during handling (Table 5.6), a material loss of 0.5% was assumed.

Table 5.2: Overview of the experimental in- and output flows for the retrieval

Flow		Unit	Trawl nets (Sassnitz)	Gillnets (Ahlbeck)
Input				
DFG_0		kg	2951	1745
		%	**100**	**100**
Big bags		kg	23.5	13.4
		%	0.796	0.768
Output				
Total		kg	2960	1750
		%	100.3	100.3
Fish (Organics)		kg	0	60
		%	**0**	**3.44**
DFG_1 + big bags		kg	2960	1690
		%	100.3	96.8
	Big bags	kg	23.5 [a)]	13.4 [a)]
		%	0.796	0.768
	DFG_1	kg	2937 [a)]	1677 [a)]
		%	99.5	96.1
Material loss (Minerals)		kg	14.8 [b)]	8.73 [b)]
		%	**0.5**	**0.5**

Factor: 4:500

a) The value assumes a ratio of 40 big bags to 5000 kg of DFG.
b) The value assumes a material loss of 0.5%.

The values in Table 5.2 were used to establish the DFG composition (Table 4.1). As part of this, the separated fish was attributed to the organic waste fraction and the material loss to the mineral waste (Appendix A).

5.1.2 Scale-up

Suitability for large-scale operations

The examined retrieval has shown to successfully remove large quantities of DFG. Therefore, it can be directly applied to large-scale operations. It is assumed that residues which drop on the floor are collected and added to the big bags which means that no material loss occurs.

Technology selection

The vessels used for the retrieval are modelled as a small shipping vessel using a diesel engine.

5.1.3 Life cycle inventory

Included activities

The retrieval comprises of three main activities, including the (1) vessel and diving operation, the (2) removal of fish and the (3) size reduction and storage (Figure 5.2). For the life cycle inventory [LCI] only the vessel operation is considered, because the other processes are carried out manually and are not expected to significantly contribute to potential environmental impacts.

Previous activities such as the identification of the DFG location and planning are not included. Also, the release and further treatment of fish is not evaluated within this thesis (Figure 5.2).

Included flows

Based on the retrieval experiments and the available literature, the following material flows were considered: (1) the time at Sea, the initial DFG_0 and the output DFG_1 as well as the live and dead fish, (2) the boat paint and antifouling and their emissions to water and (3) the diesel and lubricant oil consumption and their corresponding air emissions (Figure 5.2). The production of big bags and oxygen for the divers was not included (Figure 5.2).

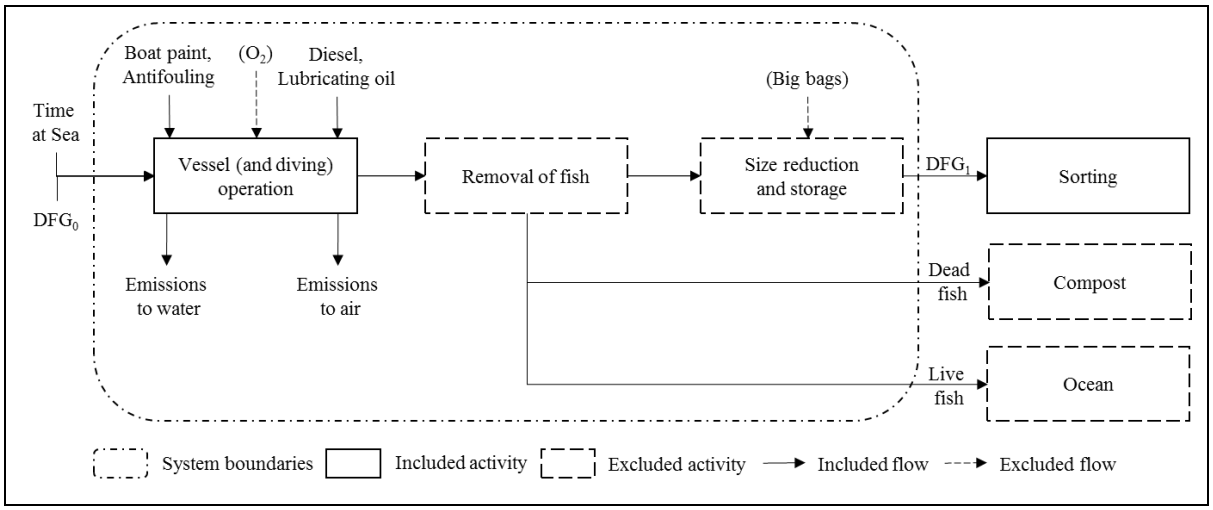


Figure 5.2: LCI scope for the retrieval process

System in- and output flows

The time at Sea was calculated as 9.56 h/t for Sassnitz and as 23.0 h/t for Ahlbeck (Table 5.3). The calculation is based on the joint retrieval and cruise time as well as the dry weight for DFG. It assumes a distance to the harbour of 50 km for Sassnitz and of 60 km for Ahlbeck. The speed of the fishing vessel was estimated as 15 km/h and the missing retrieval time from the 22nd September was assumed as 5 hours. It was estimated that the diving vessel spent approximately

half the time of the fishing vessel at Sea. To obtain the dry weight, the assumed water content of 3% was excluded from the retrieved DFG_0 (Table 5.2).

Table 5.3: Calculation of the total time at Sea

Retrieval Site	Date 2016	Fishing vessel					Diving Vessel Factor [%]	Total		
		Distance [km]	Speed [km/h]	Cruise [hh:mm]	Retrieval [hh:mm]	Time [h]		Time [h]	Dry Weight [t]	Effi-ciency [h/t]
Sassnitz	13.07	2x50 [a)]	15 [b)]	6:40	2:55	18.25	50%	27.4	2.86 [d)]	9.56
	19.07	2x50 [a)]	15 [b)]	6:40	2:00					
Ahlbeck	13.09	2x60 [a)]	15 [b)]	8:00	5:00	26	50%	39.0	1.69 [d)]	23.0
	22.09	2x60 [a)]	15 [b)]	8:00	5:00 [c)]					
Average										16.3

[a)] The Value is estimated based on Figure 5.1. [b)] The value assumes a typical speed for fishing vessels of approximately 8 knots. [c)] The value is based on the retrieval time from the 13th September 2016. [d)] The value excludes an assumed water content of 3% from DFG_0.

The modelled DFG input represents the functional unit of 1000 kg dry DFG (Table 5.4).

The fish output is calculated by multiplying the fish content in DFG with its corresponding separation efficiency. A separation efficiency of 100% was assumed, because only a negligible amount of remaining fish could be detected during sorting experiments (Chapter 5.2.1). Like the experimental retrieval, one third of the separated fish was assumed to be alive and the rest was assumed to be dead. This resulted in the modelled output of 11.5 kg for live fish and of 22.9 kg for dead fish in gillnets (Table 5.4). For trawl nets no fish was removed because the DFG composition did not contain fish. The DFG output of 1000 kg for trawl nets and of 966 kg for gillnets was calculated based on mass balance (Table 5.4).

Table 5.4: In- and output flows for the retrieval

Name	Mixed DFG	Unit	Trawl nets	Gillnets	Background data
Input					
Time at Sea	16.3	h	9.56	23.0	Calculated in Table 5.3
DFG_0	1000	kg	1000	1000	FU
Output					
Fish	17.2	kg	0	34.4	Separation efficiency = 100%
Dead fish	11.5	kg	0	22.9	Dead fish = 2/3,
Live fish	5.73	kg	0	11.5	Live fish = 1/3
DFG_1	983	kg	1000	966	Calculated based on mass balance

Ancillary products and energy

The diesel consumption of the fishing vessel was estimated as 35L/h (K. Neumann, personal communication, 19 July 2017). This accounts for 29.1 kg/h when a density of 0.832 kg/L (Thinkstep, 2019) is assumed.

The consumption of lubricant oil, boat paint and antifouling are provided in other LCAs as a measure of fish caught (Villanueva-Rey et al., 2017). For a conversion to an hourly basis, information on the catch rate would be required. However, this data could not be obtained so that the diesel consumption was used as a proxy instead. This is appropriate because a direct link between the fuel consumption and the catch rate can be expected.

As an example, the lubricant oil was reported to account for 1.97 kg for 351 kg of diesel (Villanueva-Rey et al., 2017). Thus, it was estimated that for one hour of retrieval, 0.163 kg of lubricant oil is needed (Table 5.5). The remaining data for the boat paint and antifouling was adopted from Villanueva-Rey et al. (2017) in the same way (Table 5.5).

Emissions to air

The carbon dioxide emissions are calculated as 3179 g/kg fuel based on mass conversation (European Commission, 2002). The calculation assumes a carbon content of 86.7% in marine diesel and a complete combustion into CO_2 (Ibid). This means that 92.6 kg of carbon dioxide are emitted per hour (Table 5.5).

The sulphur dioxide emissions were determined as 2 g/kg fuel. For this, a sulphur content of 0.1% in fuel (European Environment Agency [EEA], 2016a) and a full conversion into sulphur dioxide were assumed (EEA, 2016b). Given an hourly diesel consumption of 29.1 kg, a sulphur dioxide emission of 58.2 g/h must be expected (Table 5.5).

Other airborne emissions are based on the TIER 1 emission factors from the European Monitoring and Evaluation Programme [EMEP] / European Environment Agency [EEA] air pollutant emission inventory guidebook. For example, nitrogen oxides are reported to account for 78.5 kg per 1000 kg of diesel (EEA, 2016a). Consequently, 2.29 kg of nitrogen oxides would be emitted per hour during the retrieval (Table 5.5).

Emissions to water

Following the assumptions from Hospido and Tyedmers (2005) the emissions to water have been estimated as 66.6% of the boat paint and antifouling materials (Table 5.5).

Table 5.5: Life cycle inventory for 1h retrieval operation

Name	Value	Unit	SD	BU	Pedigree Score	Background data	
Ancillary products							
Diesel	29.1	kg	1.30	1.05	4,5,1,1,1	35 L/h; 0.832 kg/L	Neumann 2017; Thinkstep 2019
Lubricant oil	0.163	kg	2.28	1.05	5,5,2,1,5	1.97 kg/351 kg fuel	Villanueva-Rey et al. 2017
Boat paint							
Xylene	2.93	g	2.28	1.05	5,5,2,1,5	35.3 g/351 kg fuel	
White spirit	29.3	g	2.28	1.05	5,5,2,1,5	353 g/351 kg/fuel	
Cobalt	19.1	mg	2.28	1.05	5,5,2,1,5	230 mg/351 kg fuel	
Antifouling							
4-methyl-2-pentanone	0.556	g	2.28	1.05	5,5,2,1,5	6.70 g / 351 kg fuel	
Xylene	4.64	g	2.28	1.05	5,5,2,1,5	55.9 g / 351 kg fuel	
White spirit	64.9	mg	2.28	1.05	5,5,2,1,5	782 mg / 351 kg fuel	
Ethyl benzene	1.29	g	2.28	1.05	5,5,2,1,5	15.6 g / 351 kg fuel	
Ethanol	0.556	g	2.28	1.05	5,5,2,1,5	6.70 g / 351 kg fuel	
Copper oxide	12.9	g	2.28	1.05	5,5,2,1,5	156 g / 351 kg fuel	
Zinc oxide	6.49	g	2.28	1.05	5,5,2,1,5	78.2 g / 351 kg fuel	
Emissions to air							
Carbon dioxide	92.6	kg	1.62	1.05	4,5,5,1,1	3179 g / kg fuel	EC 2002
Sulphur dioxide	58.2	g	1.3	1.05	4,5,2,1,1	2 g / kg fuel	EEA 2016a, EEA 2018
Nitrogen oxides	2.29	kg	2.73	1.5	5,5,5,1,5	78.5 kg / t fuel	EEA 2016a
Carbon monoxide	0.215	kg	6.38	5	5,5,5,1,5	7.4 kg / t fuel	
NMVOC	81.5	g	2.73	1.5	5,5,5,1,5	2.8 kg / t fuel	
Particulates	43.7	g	2.73	1.5	5,5,5,1,5	1.5 kg / t fuel	
Lead	3.79	mg	6.38	5	5,5,5,1,5	0.13 g / t fuel	
Cadmium	0.291	mg	6.38	5	5,5,5,1,5	0.01 g / t fuel	
Mercury	0.874	mg	6.38	5	5,5,5,1,5	0.03 g / t fuel	
Arsenic	1.16	mg	6.38	5	5,5,5,1,5	0.04 g / t fuel	
Chromium	1.46	mg	6.38	5	5,5,5,1,5	0.05 g / t fuel	
Copper	25.6	mg	6.38	5	5,5,5,1,5	0.88 g / t fuel	
Nickel	29.1	mg	6.38	5	5,5,5,1,5	1 g / t fuel	
Selenium	2.91	mg	6.38	5	5,5,5,1,5	0.1 g / t fuel	
Zinc	34.9	mg	6.38	5	5,5,5,1,5	1.2 g / t fuel	
PCB	1.11	mg	4.19	3	5,5,5,1,5	0.038 mg / t fuel	
Hexachloro-Benzene	2.33	ug	4.19	3	5,5,5,1,5	0.08 mg / t fuel	
PCDD/F	3.79	ng	4.19	3	5,5,5,1,5	0.13 ug / t fuel	
Emissions to water							
4-methyl-2-pentanone	0.371	g	3.95	3	5,5,2,1,5	2/3 of input	Hospido and Tyedmers 2005
Xylene	5.04	g	3.95	3	5,5,2,1,5		
White spirit [a)]	19.6	g	3.95	3	5,5,2,1,5		
Ethyl benzene [a)]	0.863	g	3.95	3	5,5,2,1,5		
Ethanol	0.371	g	3.95	3	5,5,2,1,5		
Copper oxide	8.63	g	6.10	5	5,5,2,1,5		
Zinc oxide	4.33	g	6.10	5	5,5,2,1,5		
Cobalt	12.7	mg	6.10	5	5,5,2,1,5		

[a)] Not modelled, because inventory data is not available in Ecoinvent.

5.1.4 Critical aspects

Total time at Sea

The experiments suggest that it takes between 9.56 and 23.0 hours to retrieve 1000 kg of DFG. As the values refer to the dry weight, they also assume a water content of 3% in the DFG output. However, it is possible that the actual water content in the DFG output was much higher. In case a water content of 25% was contained, the retrieval time would increase to 12.4 and 29.8 h/t. While the effect of a higher water content on the transport process was investigated (Chapter 10.3), this was not done for the retrieval process. This is because the retrieval process is identical in all scenarios so that changes would not affect the overall results.

Apart from the dry weight, the retrieval time also depends on the distance to a specific site and its degree of pollution. If a similar level of pollution is assumed across the Baltic Sea, the data can be compared with previous studies. WWF Poland (2015) spent approximately 7800 hours at Sea to retrieve 122 tonnes of DFG. This equals to a retrieval rate of 64 h/t. In other words, it took approximately 2-7 times longer to retrieve DFG. However, this can be explained because for the retrieval near Poland a blind search was used. This means that the exact location of the DFG was unknown so that a much larger area had to be covered to find DFG. Clearly, it is more efficient to specifically target known DFG locations as described in this thesis.

To obtain more DFG locations in the future, WWF Germany is planning to equip a smaller vessel with side scan sonar to search for DFG as part of its normal shipping activity. Also, a mobile phone application was recently developed to encourage divers, fishers and other stakeholder to share DFG locations.

To reduce the time at Sea, a single vessel with a strong winch and a diving platform could be used. Also, different DFG locations could be combined during a single trip to reduce the number of cruises to the harbour. Therefore, it can be expected that the total time at Sea will be decreased in the future.

Fish output

The experiments suggest that DFG gillnets capture 3.4% of fish and that DFG trawl nets capture no fish at all. This is because trawl nets were formed as a bundle and tied to the seafloor whereas gillnets were open and partially floating in the water column (Stolte, 2019a). However, as only two retrievals were examined no general relationship between the gear type and the fish content should be assumed. Still, the results are in line with the literature which classifies

trawl nets as less likely and gillnets as the most common gear type for ghost fishing (Huntington, 2016). In other words, trawl nets represent a best- and gillnets a worst-case.

In a mixed DFG composition with an equal amount of trawl and gillnets an average fish content of 1.7% would be expected. However, this is still 4 times larger than in a previous study by WWF Poland (2015) who found 543 kg of fish in 122t of DFG. This may be explained by a different ratio of the gear type and their condition. As, the value lies within the modelled best- and worst-case no further sensitivity analysis on the fish content is required.

The separation efficiency for fish was modelled as 100%. This was justified because almost all fish could be removed during the retrieval. However, such a careful fish removal made the process very time consuming. To make the process faster, only live fish could be removed and rescued in the future. In fact, not removing the dead fish would increase the organic carbon content in DFG and thus potentially benefit a thermal treatment. However, it would also cause additional contamination and an undesirable smell during the storage, transport and pre-treatment which is why this option was not further evaluated.

5.2 Sorting

5.2.1 Experiment

Context and aim

Within MARELITT Baltic a DFG sorting experiment was conducted by Vecoplan at their industrial test facility in Bad Marienberg. This had the aim to get a better understanding of the material composition and to remove contaminants for further processing.

Materials and setup

In March 2017, Vecoplan received 4 big bags of trawl nets from Sassnitz and 9 big bags of gillnets from Ahlbeck. In general, two types of sorting were conducted – a detailed fine sorting and a faster rough sorting. For the fine sorting 3 big bags of trawl nets and 1 big bag of gillnets were used whereas the remaining material was rough sorted. For both types of sorting 3-4 people, a forklift, crane, a 1000L tank filled with water, pincers, knifes, wire cutters, and an angle grinder were deployed.

Process description

Before sorting the weight of each big bag was determined. One big bag with trawl net material showed a very high mud contamination so that a pre-washing was required. For the pre-washing the material was mounted on a crane and transferred into a water bath where it was manually stirred. After removing the material from the water bath, the actual sorting started.

During fine sorting, the DFG was positioned with a crane in an accessible upright position (Figure 5.3). Manual cutting equipment was used to remove contaminants which were grouped into the following six fractions: (1) metal, (2) wood, (3) stones, (4) textiles, (5) mussels and (6) others. Depending on the quality and type of the material the remaining rope and net material was further divided into ropes, clean nets and dirty nets. However, this was only possible for trawl net material as gillnet material was too entangled to separate further. Ultimately, all waste fractions were weighed.

Figure 5.3: Positioning of the material in preparation of the fine sorting

During rough sorting the DFG was also positioned upright by a crane to stretch the material and to better detect metal pieces. An angle grinder was used to remove the large metal pieces which would have otherwise hampered the shredding process. Afterwards, the metal fraction and the remaining big bags were weighed.

Technical challenges and process data

The fine-sorting of 100kg took approximately 3.5 - 4.5 person hours. As such the process was very time consuming and not appropriate to treat large quantities of DFG. Furthermore, there was a notable wear on the cutting tools which resulted in blunt blades that had to be replaced frequently.

The rough-sorting was much faster than fine-sorting because only large metal pieces were removed. It took approximately 0.2 - 0.4 person hours to sort 100 kg of DFG. Although rough sorting was faster, it also increased the level of contamination for the subsequent waste treatment processes. During the operation DFG got entangled in the angle grinder.

Output analysis

The cumulative weight of the three big bags containing trawl nets was 450 kg before fine sorting. After fine-sorting, ropes and nets accounted for 264 kg (Table 5.6), of which 62 kg were ropes, 52 kg clean nets and 150 kg dirty nets. The rest comprised of 41 kg of metals (Table 5.6), 37 kg of stones, 21 kg of mussels and 4 kg of textiles and wood each. The category other had a total weight of 20 kg including items such as fire hoses, cables, shoes, a balloon, plastic food wrapping, recreational fishing gear and an oxygen tank.

			
Shoe insole	Balloon	Recreational gear	Oxygen tank

Figure 5.4: Examples of items found in DFG

The one big bag with gillnet material had a total weight of 85 kg before fine sorting. From this 66 kg were classified as ropes and nets, 2 kg as mussels and 15 kg as other. The category other contained 3 bottles with liquid, a minor amount of dead fish and some lead lines.

The single big bag with trawl nets used for rough sorting had a weight of 277 kg. The weight of the big bag was 2 kg so that 275 kg of the material was DFG (Table 5.6). From this 4 kg were separated in form of large metal pieces, particularly chains.

The total weight of the eight big bags with gillnet material was 783 kg before rough sorting. From this 13 kg accounted for big bags and 770 kg for DFG (Table 5.6). The output metal fraction mainly comprised of anchors and chains and had a total weight of 53 kg.

The water content in the weighted DFG fractions was not determined. However, all materials had time to dry prior to the weighting. Thus, following previous assumptions (Chapter 5.1.1), a water content of 3% is assumed for the DFG output weights.

Waste composition

An overview of the in- and output flows is presented in Table 5.6. The average weight of a big bag from rough sorted DFG was approximately 1.6 kg. The same value was assumed for the fine sorted DFG. All output fractions other than metal and DFG were summarised as a residual fraction. The difference between the in- and output flows was calculated for the fine sorted DFG which revealed a material loss of 12.2% for gillnets and of 0.5% for trawl nets. The large difference can be explained by the pre-washing step that was applied to a subset of the fine sorted trawl nets. For the rough sorted materials, no pre-washing took place so that a material loss of 0.5% was assumed. Ultimately, the DFG output weight for the rough sorted materials was calculated based on the mass balance.

To establish the dry DFG composition (Table 4.1), the separated residual fraction and material loss were directly allocated to the mineral waste and the separated metal was accounted as steel (Appendix A).

Table 5.6: Overview of the experimental in- and output flows for the sorting

Flow	Unit	Trawl nets			Gillnets		
		Fine sorted	Rough sorted	**Total**	Rough sorted	Fine sorted	**Total**
Input							
DFG_1 + big bags	kg	450	277	727	783	85	868
	%	101.1	100.7	100.9	101.9	101.7	101.7
Big bags	kg	4.8 [a)]	2	6.8	13	1.6 [a)]	14.6
	#	3	1	4	8	1	9
	%	1.1	0.7	0.9	1.9	1.7	1.7
DFG_1	kg	445	275	720	770	83.4	853
	%	100	100	100	100	100	100
Output							
Metal (Steel)	kg	41	4	45	53	0	53
	%	9.2	1.5	**6.2**	6.9	0	**6.2**
Residual fraction (Minerals)	kg	86	0	86	0	17	17
	%	19.3	0	**11.9**	0	20.4	**2.0**
Material loss (Minerals)	kg	54.2 [b)]	1.38 [d)]	55.6	3.85 [d)]	0.4 [b)]	4.25
	%	12.2	0.5	**7.7**	0.5	0.5	**0.5**
DFG_2	kg	264 [c)]	270 [b)]	534	713 [b)]	66	779
	%	59.3	98.0	**74.1**	92.6	79.1	**91.3**

[a)] The value is calculated as initial weight of the big bags minus 1.6 kg per big bag. [b)] The value is calculated based on mass balance. [c)] This fraction is further divided into ropes (62 kg), clean nets (52 kg) and dirty nets (150 kg) [d)] The value is estimated at 0.5% of the input weight, based on gillnet fine-sorting experiments.

5.2.2 Scale-up

Suitability for large-scale operations

The rough sorting is suitable for a large-scale production and thus used as basis for the LCI.

5.2.3 Life cycle inventory

Included activities

The activities during rough sorting are the big bag removal, a pre-washing and the metal removal (Figure 5.5). For the LCI only the metal removal is considered.

The pre-washing is excluded because the manual stirring process could be directly conducted as part of the retrieval. If considered, a more sophisticated technology with an individual wastewater treatment would be required (Figure 5.5).

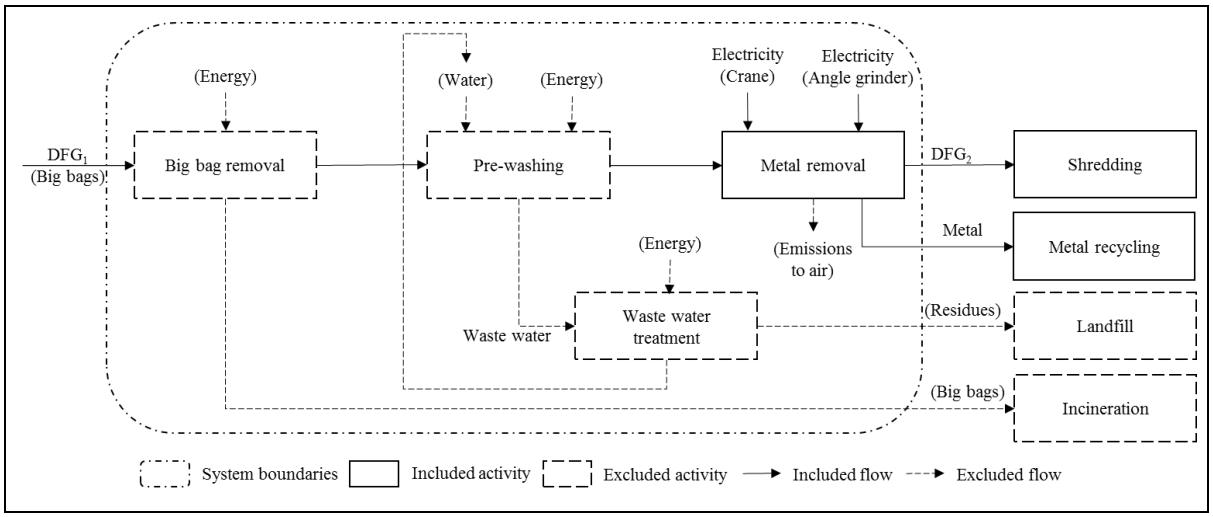


Figure 5.5: LCI scope for the Sorting

Included flows

Based on the experimental data and machine specifications the following flows for the sorting are included: (1) the DFG_1 input, the DFG_2 output and the metal output as well as (2) the electricity consumption of the crane and the angle grinder. Airborne emissions from the metal removal are not included but may be considered in the future (Figure 5.5).

In- and output flows

The DFG input is assumed to equal the DFG output from the retrieval (Table 5.4).

The metal output is calculated by multiplying the ferrous metal content in DFG with the metal separation efficiency from rough sorting. The separation efficiency is determined as 79.2% in Table 5.7. For the calculation it was assumed that only rough sorting (Chapter 5.2.1) and rough shredding (Chapter 5.3.1) contributed to the metal output. The rough sorting separated 57 kg of metal (Table 5.6) whereas 15 kg were removed during the rough shredding (Table 5.13).

Table 5.7: Separation efficiency for metal

Metal	Sum [kg]	Background data	Separation [%]
Total	72		100
Rough Sorting	57	Table 5.6	79.2
Rough Shredding	15	Table 5.13	(20.8)

The initial metal content was established as 7.4% and 7.3% for trawl and gillnets respectively (Table 4.1). At a separation efficiency of 79.2%, 59.0 kg of metal from trawl nets and 57.7 kg of metal from gillnets will be separated during the rough shredding (Table 5.8). This means that approximately 941 kg of DFG from trawl nets and 908 kg of DFG from gillnets remain.

Table 5.8: In- and output flows for Sorting

Name	Mixed DFG	Unit	Trawl nets	Gillnets	Background data
Input					
DFG_1	983	kg	1000	966	Table 5.4
Output					
Metal	58.4	kg	59.0	57.7	Separation efficiency = 79.2%
DFG_2	924	kg	941	908	Calculated based on mass balance

Ancillary products and energy

The electricity consumption of the crane is determined as 0.045-0.324 Wh/kg dry DFG (Table 5.9). For this, a crane with a 5t hoist engine and power output of 0.4-2.9 kW (Hitachi Ltd., n.d.) as well as a process time of 5 min for 770 kg DFG with a water content of 3% were assumed.

The electricity consumption of the angle grinder was calculated as 0.127-0.225 Wh/kg dry DFG (Table 5.9). It was estimated that a 4.5-inch angle grinder with a typical power range of 0.57-1.01 kW (Screwfix Direct, 2018) requires 10 min to process 770 kg of DFG with an assumed water content of 3%.

Table 5.9: Life cycle inventory for sorting 1kg of dry DFG

Name	Value	Unit	Min	Max	Background data
Ancillary products					
Electricity (crane)	0.184	Wh	0.045	0.324	0.4-2.9 kW, 5 min, 770 kg, 3% water
Electricity (angle grinder)	0.176	Wh	0.127	0.225	0.57-1.01 kW, 10 min, 770 kg, 3% water

5.2.4 Critical aspects

Metal output

The sorting's metal output depends on the metal separation efficiency which was nearly 100% during fine sorting and 79.2% during rough sorting. A higher separation efficiency is desired

because it reduces the wear on the shredder. In the future, the separation efficiency of the rough sorting may be increased by applying a metal detector for the identification of large metal pieces. It could be coupled with a guillotine to automatically cut out the metal pieces.

The removed metal is assumed to be recycled (Chapter 8.1). However, it is also possible that anchors and metal chains are returned to the fishers for a reuse. This will depend on the condition of the removed metal products.

DFG output

Apart from large metal pieces, other materials such as floaters or lead lines could be recovered for a reuse. This would reduce the contamination of possibly toxic elements and therefore benefit the subsequent waste treatment processes. However, during the experiments the lead lines were heavily entangled with the remaining DFG and an effective recovery seemed not possible. In the future, a recovery of the valuable materials may be achieved, for example, when more experienced fishers assist with the sorting.

As part of a mechanical recycling other processing steps may be implemented to improve the quality of the sorting output. For example, polymers could be separated based on their mechanical properties and grouped according to their colours. Plastix Global estimated that approximately 50% of the polymers from the considered DFG would need to be removed due to their inferior mechanical properties (H.A. Kristensen, personal communication, 21 November 2017). This fraction could be reduced when a special master batch is applied during the extrusion. Still, a sorting of DFG material would take 10-12 time longer than EOL fishing gear which renders it uneconomic (Ibid).

Electricity consumption

The modelling of manual sorting or dismantling processes are typically neglected in other LCAs, because they do not significantly contribute to the environmental impacts (Gu et al. 2017; Jenseit et al., 2003). In fact, even automated sorting equipment such as ballistic separators or near infrared scanner, with a typical electricity consumption of 1.2-1.8 kWh/t (Shonfield, 2008), will not significantly contribute to the environmental impacts because their power consumption is much smaller than for example shredding.

5.3 Shredding

5.3.1 Experiment

Context and aim

Within MARELITT Baltic, a shredding experiment was conducted by Vecoplan at their technology centre. The aim of the shredding was to investigate the machine's ability to handle fibrous DFG material and to prepare smaller particles for the next waste treatment steps.

Materials and setup

The sorted net and rope fractions served as input for the shredding. This contained approximately 264 kg of fine sorted trawl nets, 66 kg of fine sorted gillnets, 270 kg of rough sorted trawl nets and 713 kg of rough sorted gillnets (Table 5.6).

The shredding was conducted in two stages. For the initial rough shredding the uniaxial VAZ 2000 MNFT was equipped with a 120mm screen and a rotor containing 48 cutting crowns of 80 x 80mm. The subsequent fine shredding was conducted with the uniaxial VAZ 1600 MXLT which was equipped with a rotor containing 72 cutting crowns of 40 x 40mm. For the fine and rough sorted DFG materials a 20 mm and 30 mm screen were used respectively. Apart from the shredders, also a forklift, a conveyor belt, a magnet and several big bags were used.

Process description

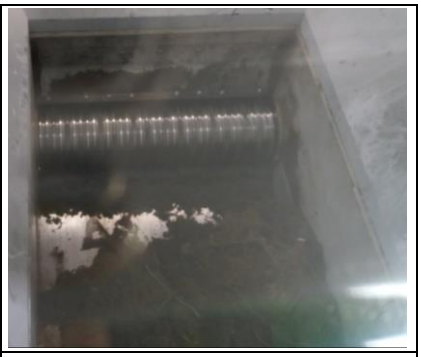
For the rough shredding, the DFG was charged into the shredder and pushed against the rotor to cut it in between the cutting crowns and counterknifes Figure 5.6). Then, the shredded material was pressed against the specified screen Figure 5.6) so that smaller particles fell onto the conveyor belt. The conveyor belt transported the shredded material to the magnet to separate the magnetic materials from the remaining DFG, before the fractions were collected in big bags. The DFG output from trawl and gillnets were separately collected whereas their corresponding metal fractions were collected together. All output fractions were weighed.

For the fine shredding, the big bags were removed prior to the material charging. The shredding process, the magnetic separation and the material collection took place in the same way as for rough shredding. At the end of the process all output fractions were visually assessed, and the weight was determined for the rough sorted materials. Samples of the rough sorted gillnet material were analysed by PreZero, Clean Carbon Conversion and the RWTH Aachen University as well as the Hochschule Magdeburg.

Throughout rough and fine shredding, the power output of the shredding machine was recorded for the rough sorted materials and the fine sorted gillnets. To reduce the spreading of lead, the trawl nets were processed before the gillnet material. Fine sorted material did not require a rough shredding due to their lower level of contamination.

Rotor with cutting crowns

Screen



View into operating shredder

Figure 5.6: Visualisation of the shredding process

Technical challenges and process data

During rough shredding an automatic back turning mode to resolve blockages was frequently activated which indicates a high level of hard material contamination. After the process, the knifes and cutting crowns showed notable wear and at least one counterknife was ejected completely. At this level, the blades were estimated to last 2-4 days of operation before a replacement would be needed (U. Kramer, personal communication, 13 November 2017). To decrease the wear on the machine, a higher separation efficiency for large metal pieces and possibly other hard materials is required during sorting for which a metal detector could be used.

The fine shredding worked well. The back-turning mode was rarely activated so that the process operated without interference.

During rough and fine shredding no dust or heat development could be noted. However, there was concern that particularly the fine shredding would free the lead from the lead lines and thus negatively influence the subsequent waste treatment processes. Therefore, a density separation was attempted with the rough shredded gillnet material. This had no effect however, because the entangled lead lines caused the complete fraction to sink. Therefore, a fine shredding to approximately 20-30mm was necessary.

The data on the power output at full load is provided in Table 5.10. The rough shredding had an average power output of 44.4 - 50 kW whereas fine shredding spent approximately 60 kW.

The higher energy requirement for fine shredding may be explained by the smaller screen size and a possibly higher cutting effort for smaller particles.

Table 5.10: Power output data for shredding

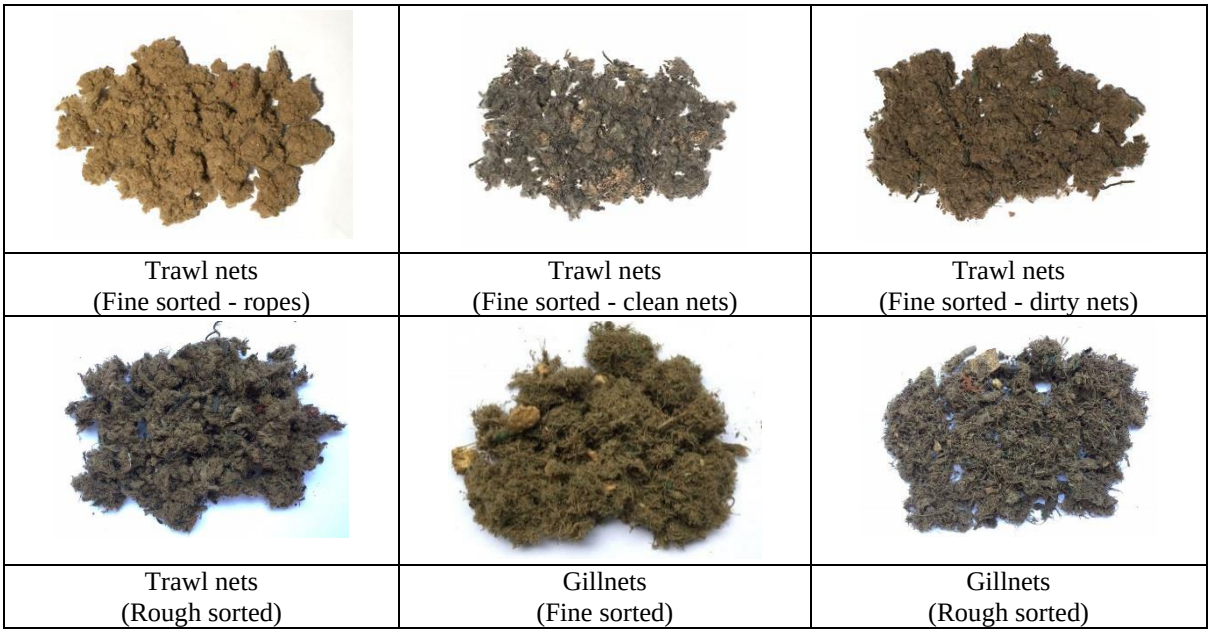
Process	Flow	Unit	Trawl nets				Gillnets	
			Fine sorted			Rough sorted	Fine sorted	Rough sorted
			Ropes	Clean	Dirty			
Rough Shredding	Process time	min	☒	☒	☒	10	☒	18
	Average power	kW	☒	☒	☒	44.4	☒	50
Fine Shredding	Process time	min	-	-	-	20	33	30
	Average power	kW	-	-	-	60	16	60

☒ Not applicable, because process was not conducted. ⊟ No data available.

Output analysis

The big bag with rough shredded trawl nets had a weight of 243 kg and the big bag with rough shredded gillnets weighed 706 kg. In both cases the approximately 100 – 120 mm long output fibres were heavily entangled. The big bag with rough shredded magnetic materials had a weight of 16 kg.

The fine shredded DFG output fractions are presented in Figure 5.7. The measured weights of the individual big bags with fine shredded material were 219 kg for the rough sorted trawl nets and 678 kg for the rough sorted gillnets. The amount of separated magnetic materials was less than 100g and not considered for further analysis.



Trawl nets (Fine sorted - ropes)	Trawl nets (Fine sorted - clean nets)	Trawl nets (Fine sorted - dirty nets)
Trawl nets (Rough sorted)	Gillnets (Fine sorted)	Gillnets (Rough sorted)

Figure 5.7: Fine shredded DFG output fractions

The laboratory results of the rough sorted gillnets are presented in Table 5.11 and Table 5.12. Interestingly, the water and ash content as well as the lower heating values (LHV) showed large

differences ranging from 1.9–25.0%, 30.9–61.3% and 4.72–27.507 MJ/kg respectively (Table 5.11). This can be explained, because the samples were not taken at the same date or from the same location within the big bag.

Table 5.11: Characterisation of the rough sorted gillnets after fine shredding

Reference	Water [%]	Ash [a)] [%]	LHV [a)] [MJ/kg]	C [b)] [%]	O [b)] [%]	H [b)] [%]	N [b)] [%]	F [b)] [%]	Cl [b)] [%]	S [b)] [%]
Schurig, 2017a	18.9	61.3	12.797	-	-	-	-	<0.01	0.43	0.09
Weißbach and Gerke, 2018	1.9	45	-	-	-	-	-	-	-	-
Hee, Horst and Quicker, 2018	3.5	58.0	4.720	29.1	-	4.4	3.5	-	0.3	0.05
Rupert, Haupt and Küttel, 2017	25.0	30.9	27.507	46	8.5	8	7.1	-	-	0.06

⊟ No data available. [a)] The values are based on a wet basis. [b)] The values are based on a dry basis.

The samples for the RWTH Aachen (Hee, Horst and Quicker, 2018) and the Hochschule Magdeburg (Weißbach and Gerke, 2018) were directly taken after the experiment. As such they closely represent the actual water content in the above measured DFG output for which a typical value of 3% can be assumed. The higher water content in the other samples is presumably linked to rainfall that entered the big bag after the shredding experiment while it was stored openly outside the Vecoplan facility.

The different ash content and heating values can be explained by the heterogeneity of the DFG material. In fact, a thorough mixing of the DFG material did not take place which allowed high density materials such as sediments and lead to accumulate at the bottom of the big bag (Chapter 7.2.1). Although a sample from the bottom of the big bag was not taken, the example illustrates that the bulk characteristics may not be appropriately reflected. As such the presented information must be treated with care. A sensitivity analysis on the waste composition was conducted in Chapter 10 to address this uncertainty.

The results of the heavy metal concentrations in rough sorted gillnet is presented in Table 5.12. The most notable result is the lead content which accounts for 2.74% of the input weight. However, it must be pointed out that this value is only representative for the sample material at the top of the big bag, where lead did not accumulate. This top fraction was then given to a density separation (Chapter 5.3.1) during which approximately 95% of the lead was separated (Table 6.6). The washing of this top fraction further reduced the lead content to below 400 ppm (Table 6.11). Shredded DFG gillnets at the bottom of the big bag had a lead content of approximately 32.4% (Table 7.6). Due to the large difference the trials could not be directly compared. Therefore, an average waste composition was calculated based on both trials (Appendix A) to define the functional unit for the LCA.

Table 5.12: Heavy metal concentration of the rough sorted gillnets after fine shredding

Unit	As	Pb	Cd	Cr	Cu	Ni	Hg	Tl	Sb	Sn	Co	Mn	V
mg/kg	3.0	27400	0.479	33.2	147	19.2	<0.5	<0.5	67.7	113	171	604	4.2

Table adapted from: Schurig (2017a)

Waste composition

The collected mass data is summarised in Table 5.13. The DFG input is based on the DFG output from sorting (Table 5.6). For the output fractions the big bags were excluded. The metal output from rough shredding was equally attributed to gillnets and trawl nets based on their input weights. The material loss of the rough sorted materials during fine shredding was calculated as 5.5% which was also assumed for the fine sorted DFG. The other fractions were determined based on the mass balance (Table 5.13).

Table 5.13: Overview of the experimental in- and output flows for the shredding

Process	Flow	Unit	Trawl nets					Gillnets		
			Fine sorted			Rough sorted	Total	Rough sorted	Fine sorted	Total
			Ropes	Clean	Dirty					
Rough Shredding	Input									
	DFG	kg	☒	☒	☒	270 [a)]	270	713 [a)]	☒	713
		%	☒	☒	☒	100	100	100	☒	100
	Output									
	Metal (ferrous)	kg	☒	☒	☒	4.12 [b), c)]	4.12	10.9 [b), c)]	☒	10.9
		%	☒	☒	☒	1.5	**1.5**	1.5	☒	**1.5**
	Mat. loss (% comp.)	kg	☒	☒	☒	23.5 [d)]	23.5	-2.7 [d)]	☒	-2.7
		%	☒	☒	☒	8.7	**8.7**	-0.4	☒	**-0.4**
	DFG	kg	☒	☒	☒	242 [b)]	242	705 [b)]	☒	705
		%	☒	☒	☒	89.8	89.8	98.9	☒	98.9
Fine Shredding	Input									
	DFG	kg	62 [a)]	52 [a)]	150 [a)]	242 [a)]	506	705 [a)]	66 [a)]	771
		%	100	100	100	100	100	100	100	100
	Output									
	Metal (ferrous)	kg	<1	<1	<1	<1	0	<1	<1	0
		%	0	0	0	0	**0**	0	0	**0**
	Mat. loss (% comp.)	kg	3.41 [e)]	2.86 [e)]	8.25 [e)]	24 [d)]	38.5	28.0 [d)]	3.63 [e)]	31.6
		%	5.5	5.5	5.5	9.9	**7.6**	4.0	5.5	**4.1**
	DFG	kg	58.6 [d)]	49.1 [d)]	142 [d)]	218 [b)]	467	677 [b)]	62.4 [d)]	739
		%	94.5	94.5	94.5	90.1	92.4	96.0	94.5	95.9

☒ Not applicable, because process was not conducted. [a)] No material losses occurred after the previous process. [b)] The weight of one big bag is subtracted. [c)] Assumes the same metal composition for gillnets and trawl nets. [d)] The value is calculated based on mass balance. [e)] Estimated at 5.5% based on joint loss of rough-sorted trawl and gillnets.

This resulted in a negative value for the material loss during the rough shredding of gillnets. However, this material gain can be explained because trawl nets were processed immediately before the gillnets and because a thorough cleaning of the shredding machine was not possible within the industrial setting of the experiment. Consequently, some of the remaining trawl net material will have been added to the gillnets.

For the calculation of the waste composition (Table 4.1) the separated metal content was allocated to ferrous metals and the material loss was allocated across different categories to reflect the input material composition of each processing step (Appendix A).

5.3.2 Scale-up

The experimental rough and fine shredding is suitable for large-scale processing and is thus providing the modelling basis for the LCI. It is assumed that the material loss is returned to the process during an industrial processing.

5.3.3 Life cycle inventory

Included activities

The shredding entails two sequences of charging, rough shredding and magnet separation (Figure 5.8). For the LCI only the shredding processes are included. This is because the shredding process can be expected to represent the biggest environmental impact.

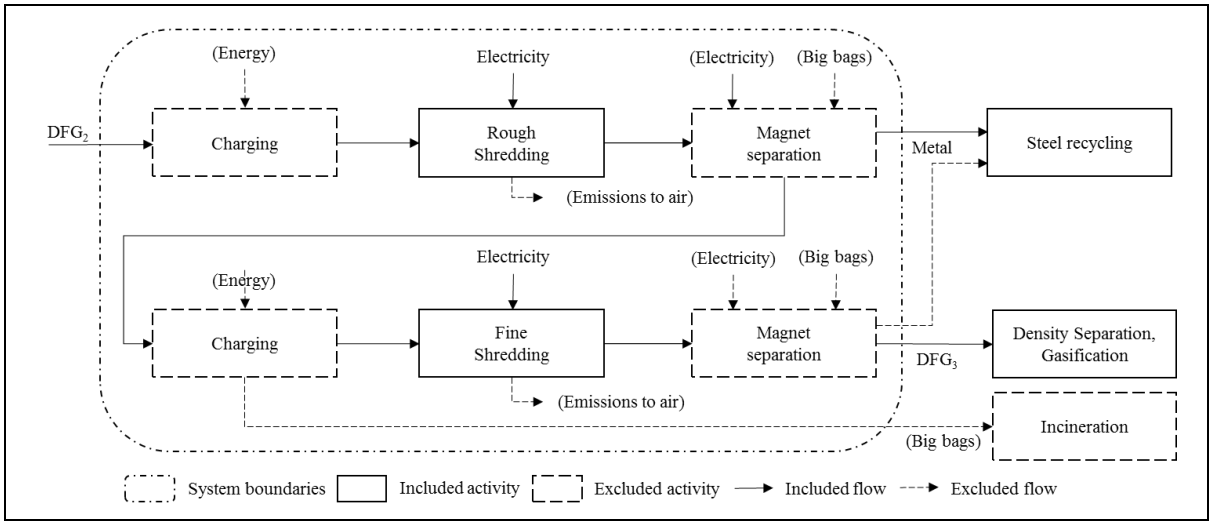


Figure 5.8: LCI scope for shredding

Included flows

The included flows are (1) the DFG_2 input, the DFG_3 output and the metal output and (2) the electricity consumption of the shredding processes. Possible emissions from shredding in form of microfibres or volatile compounds were not included due to a lack of data (Figure 5.8).

In- and output flows

The amount of DFG input is assumed to equal the DFG output from sorting (Table 5.8). The metal output is calculated by multiplying the remaining ferrous metal content in DFG with the

metal separation efficiency for shredding. The steel content after sorting is approximately 15.5 kg for trawl nets and 15.2 kg for gillnets. A steel separation efficiency of 100% was assumed. This is because two consecutive magnets are deployed which ensures that the magnetic materials are removed. In fact, the experiments indicated that one magnet may be enough to remove nearly all the magnetic material. This means that after shredding approximately 926 kg of trawl nets and 893 kg of gillnets remain for the following process steps (Table 5.14).

Table 5.14: In- and output flows for shredding

Name	Mixed DFG	Unit	Trawl nets	Gillnets	Background data
Input					
DFG_2	924	kg	941	908	Table 5.8
Output					
Metal	15.4	kg	15.5	15.2	Separation efficiency = 100%
DFG_3	909	kg	926	893	Based on mass balance

Ancillary products and energy

The electricity consumption for the rough shredding was established as 21.7-28.3 Wh/kg dry DFG and for the fine shredding as 43.9-85.2 Wh/kg dry DFG (Table 5.15). The calculation was based on the experimental data for the rough sorted DFG (Table 5.10 and Table 5.13) and an assumed water content of 3%.

Table 5.15: Life cycle inventory for the shredding of 1 kg dry DFG_2

Name	Value	Unit	Min	Max	Background data
Ancillary products					
Electricity (Rough shredding)	25.0	Wh	21.7	28.3	50.0 kW, 18 min for 713 kg, 3% water; 44.4 kW, 10 min for 270 kg, 3% water
Electricity (Fine shredding)	64.5	Wh	43.9	85.2	60.0 kW, 30 min for 705 kg, 3% water; 60.0 kW, 20 min for 242 kg, 3% water

5.3.4 Critical aspects

Metal output

The ferrous metal output is determined based on the metal separation efficiency. The assumed separation efficiency of 100% represents a best-case which may not be achieved in real life. In fact, the gasification experiment revealed the presence of magnetic materials in the residue fraction (Chapter 7.2.1) and an analysis of the screen fraction from washing revealed an iron content of 0.1% (Table 6.12). Therefore, the assumed separation efficiency is clearly overestimated. Still, the application of two magnets ensures that most ferrous metal is removed during shredding so that a nearly complete separation is justified. This in line with previous LCA

studies which assume metal separation efficiencies of 100% (Unger et al., 2017; Tunesi, Baroni and Boarini, 2016).

Electricity consumption

The total electricity consumption for the shredding of DFG lies between 65.6 and 113 kWh/t. This appears large when compared to the literature. For example, Beigbeder et al. (2019) measured an electricity consumption of 43 kWh/t for the shredding of biodegradable plastics, Shonfield (2008) reported 16-32 kWh/t for mixed plastics and Biganzoli et al. (2015) documented 26.5 kWh/t for the shredding of flat panel displays. However, a direct comparison is not fair because the material composition and form of DFG differs from the referenced studies on hard plastics. Also, the screen size of the shredding equipment is not provided which prevents a meaningful evaluation. Still, it is possible that more energy efficient shredding machines are available in Europe. This means, that less energy may be required for the shredding of DFG in the future.

The conveyor belt and the magnet were excluded from the LCI. Rigamonti et al. (2014) give an electricity consumption of 0.75 kWh/t for magnets. This represents less than 1% of the average electricity consumption of the shredding process so that an exclusion can be justified.

6 Density Separation, Washing and Drying

6.1 Density Separation

6.1.1 Experiment 1

Context and aim

Within shredded DFG several materials were identified with distinct density ranges (Table 6.1).

Table 6.1: Expected materials in DFG after shredding and their typical densities

Material	Density [g/cm3]	References
Lead	11.34	Callender, 2003
Mussels (CaCO3)	2.71	Maier and Calafut, 2001
PET	1.37-1.45	Lupo et al., 2016
Sediments	1.2 -1.3	Håkanson et al., 2013
PA6	1.13-1.16	Orasutthikul et al. 2017; Weißbach and Gerke, 2018
PE	0.91-0.94	Lupo et al., 2016
PP	0.86-0.95	Lupo et al., 2016

To obtain the desired nylon fraction for a mechanical recycling, a two-stage manual sink-float separation was conducted by Vecoplan in Bad Marienberg. In a first stage the higher density materials and in the second stage the lower density materials were removed (Figure 6.1).

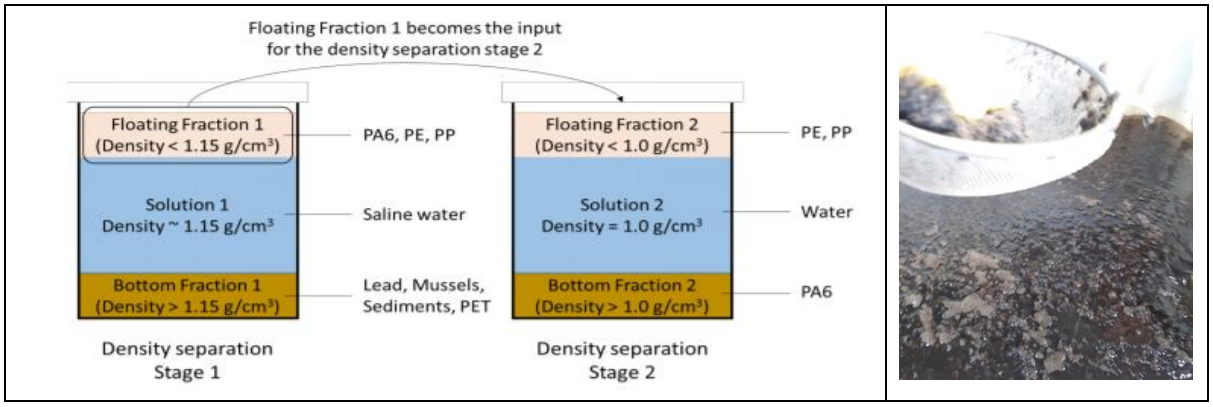


Figure 6.1: Visulisation of the density separation (left: schemata; right: actual process)

Materials and setup

For the sink-float separation approximately 62.4 kg of fine-sorted gillnets, 58.6 kg of fine-sorted trawl net ropes, 49.1 kg of fine sorted clean trawl nets and 142 kg of fine sorted dirty trawl nets were used (Table 5.6). In addition, approximately 16 kg of rough sorted trawl nets and 292 kg of rough sorted gillnets were processed.

As part of the preparation, a 1000 litre tank was filled with water and salt and stirred until a nylon test specimen was able to float. The density of the solution was estimated at 1.15 g/cm^3 (Stolte and Schneider, 2018). A second tank was filled with water having a density of 1.0 g/cm^3.

Furthermore, a cable drill, household sieves and a shovel as well as a crane, a forklift and big bags were used throughout the processing.

All materials were processed after each other. However, the clean and dirty trawl nets as well as the rough sorted DFG shared the same saline water. A density separation in plain water was not conducted for the fine sorted trawl nets, because hand trials did not yield separated fractions.

Process description

During the density separation batches of approximately 30-50 L of shredded DFG were directly loaded from the big bag into the tank with saline water. After stirring, the material was given approximately one minute to settle, before the floating fraction was skimmed off from the top. The wet floating fraction was manually squeezed, and the saline water returned to the tank.

During the second density separation the squeezed fraction was loaded into the plain water bath before stirring, skimming and squeezing was performed in the same way as before.

After each material group had been processed, the remaining suspension was released from the tanks so that the bottom fractions could be evaluated. The desired polymer fractions were loaded into big bags and hanged on a crane for drying whereas the other output fractions were stored openly.

All output fractions were visually assessed. In addition, the residual floating fractions and the bottom fraction of the fine sorted gillnets and trawl nets ropes were weighed.

Samples of the rough sorted gillnets were taken and analysed. The TU Clausthal determined the heavy metal concentrations in the saline wastewater and bottom fraction and PreZero commissioned a detailed analysis of the floating fraction from both density separations.

Technical challenges and process data

The sink-float separation worked well in principle. It was observed that with increasing time and stirring more fibres started to float. Still, the manual process was very inefficient. The separation of 100 kg of trawl and gillnets took between 8-10 person-hours. Although different sieves were tested, an increase of the overall efficiency could not be achieved (Stolte and Schneider, 2018). In the future, automated processes for this step need to be considered.

Output analysis

The visual examination revealed a dark brown residual water. The bottom fractions mainly contained sediments and a minor fraction of fibrous materials. In the bottom fractions from gillnets, pieces of lead were detected. The floating fractions comprised of mixed fibrous materials in different colours.

The weights of the floating fractions were 18kg for the fine sorted gillnets and 4kg for the rough sorted gillnets. The rough sorted trawl nets yielded less than 20g as a floating fraction so that this floating fraction was neglected for further analysis. The bottom fraction of fine sorted gillnets weighed 12 kg whereas the fine sorted trawl net rope resulted in a bottom fraction of 22 kg. The reported weights are assumed to include a water content of 3%. This is because the material fractions were given time to dry prior to the weight measurement.

The results of the wastewater and bottom fraction analysis are provided in Table 6.2. They reveal the presence of heavy metals in the rough sorted gillnets. Particularly the high lead content of 10.6-15.5% stands out.

Table 6.2: Characterisation of density separation residues from rough sorted gillnets

Sample	**As**	**Cd**	**Co**	**Cr**	**Cu**	**Fe**	**Mn**	**Ni**	**Pb**	**Zn**
	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L
Wastewater	0,18	<0,05	0,38	<0,05	0,98	2,17	5,01	<0,1	73,6	0,68
Fine bottom fraction	3,6	0,4	167	14	194	6600	490	7	105819	313
Coarse bottom fraction	2,6	0,5	90	24	381	4148	313	13	155336	406

Table adapted from: Sommer and Hoffmann, 2017

The analysis of the floating fractions from rough sorted gillnets is presented in Table 6.3. Both material fractions showed high heating values of 25.4-27.8 MJ/kg. The lead content decreased from previously 2.7% after shredding (Table 5.12) to below 0.3% in both samples. The chlorine content in DFG_4 was more than 10 times higher than in the floating fraction. However, this can be explained by the different salt content in the process water.

Table 6.3: Characterisation of the floating fractions from rough sorted gillnets

Sample	**Ash** [%]	**LHV** [MJ/kg]	**F** [%]	**Cl** [%]	**S** [%]	**Hg** [ppm]	**Tl** [ppm]	**Sb** [ppm]	**Sn** [ppm]
DFG_4 [a)]	21.92	25.434	<0.01	6.55	0.09	<0.5	<0.5	21.3	15.6
Floating fraction [b)]	11.53	27.842	<0.01	0.49	0.09	<0.5	<0.5	42.8	12
Sample	**As** [ppm]	**Pb** [ppm]	**Cd** [ppm]	**Cr** [ppm]	**Cu** [ppm]	**Ni** [ppm]	**Co** [ppm]	**Mn** [ppm]	**V** [ppm]
DFG_4 [a)]	<1	2790	1.19	18	52.2	10.4	102	359	2.24
Floating fraction [b)]	<1	2010	0.659	11.8	50.6	4.81	60.6	251	1.96

[a)] Data from: Schurig, 2017b; [b)] Data from: Schurig, 2017c

Waste composition

The in- and output flows for the density separation are summarised in Table 6.4. The input weights were directly taken from Table 5.13 or calculated based on the mass balance when not all material was used. The bottom fraction accounted for 37.5% of the output weight in trawl nets ropes and for 19.2% in fine sorted gillnets. The same values were assumed for the remaining trawl and gillnets respectively. The lead content in the bottom fraction from trawl nets was assumed to be zero because no lead could be observed. Based on the analysis presented in Table 6.2 an average lead content of 13.5% was assumed for the bottom fraction of gillnets. A material loss in form of misplaced material was estimated as 1% following visual observations. The DFG output and the residues in the bottom fraction were calculated based on the mass balance (Table 6.4).

Table 6.4: Overview of the experimental in- and output flows for the density separation

Process	Flow	Unit	Trawl nets					Gillnet		
			Fine sorting			Rough sorted	Total	Rough sorted	Fine sorted	Total
			Ropes	Clean	Dirty					
1. Density Separation	Input									
	DFG_3	kg	58.6 [a)]	49.1 [a)]	142 [a)]	16 [b)]	265	292 [b)]	62.4 [a)]	354
		%	100	100	100	100	100	100	100	100
	Output									
	Bottom Fr.	kg	22	18.5 [c)]	53.2 [c)]	6.01 [c)]	99.7	56.2 [c)]	12	68.2
		%	37.5	37.5	37.5	37.5	37.5	19.2	19.2	19.2
	Lead (Lead)	kg	0 [d)]	0 [d)]	0 [d)]	0 [d)]	0	7.58 [e)]	1.62 [e)]	9.20
		%	0	0	0	0	**0**	2.6	2.6	**2.6**
	Residues (Minerals)	kg	22 [b)]	18.5 [b)]	53.2 [b)]	6.01 [b)]	99.6	48.6 [b)]	10.4 [b)]	59.0
		%	37.5	37.5	37.5	37.5	**37.5**	16.6	16.6	**16.6**
	Material loss	kg	0.586 [f)]	0.491 [f)]	1.42 [f)]	0.16 [f)]	2.65	2.92 [f)]	0.624 [f)]	3.54
		%	1.0	1.0	1.0	1.0	**1.0**	1.0	1.0	**1.0**
	DFG_4	kg	36.0 [b)]	30.2 [b)]	87.1 [b)]	9.83 [b)]	163	233 [b)]	49.7 [b)]	283
		%	61.5	61.5	61.5	61.5	61.5	79.8	79.8	79.8
2. Density Separation	Input									
	DFG_4	kg	☒	☒	☒	9.83 [g)]	9.83	233 [g)]	49.7 [g)]	283
		%	☒	☒	☒	100	100	100	100	100
	Output									
	Floating fr. (PE/PP)	kg	☒	☒	☒	<<1	0	18	4	22
		%	☒	☒	☒	0	**0**	7.7	8.0	**7.8**
	Material loss	kg	☒	☒	☒	0.098 [f)]	~0.1	2.33 [f)]	0.497 [f)]	2.83
		%	☒	☒	☒	1	**1**	1	1	**1**
	DFG_5	kg	☒	☒	☒	9.73 [e)]	9.73	213 [e)]	45.2 [b)]	258
		%	☒	☒	☒	99.0	99.0	91.4	90.9	91.2

☒ Not applicable, because process was not conducted. [a)] The value is taken from Table 5.13. [b)] The value is calculated based on mass balance. [c)] The value assumes the same relative bottom fraction across the same material category. [d)] The value assumes a lead content of zero. [e)] The value assumes a lead content of 13.5 % for the bottom fraction. [f)] The value assumes a material loss of 1%. [g)] The value is based on the first density separation.

To determine the composition of dry DFG (Table 4.1), the separated lead was allocated to the lead, the residues were allocated to the mineral fraction and the floating fraction was allocated

to the polymer fraction (Appendix A). The material loss was allocated to the relative input composition of the density separation (Appendix A).

6.1.2 Experiment 2

Context and aim

Within MARELITT Baltic, Andritz Separation conducted a centrifugal density separation. Like the sink-float separation (Chapter 6.1.1), this had the aim to separate nylon from other DFG material. It was planned to remove the denser materials first and the lighter materials afterwards. However, due to technical challenges only the first separation took place.

Material and setup

One big bag with approximately 220 kg of rope material served as input material. Unlike previously mentioned materials, this fraction derived from a retrieval near Rügen. It was mechanically pre-cleaned at a recycling facility in Poland and shredded with a 10mm screen at the Vecoplan test center.

The density separation took place on a newly built ACZ 4-3 centrifuge which was mounted on a truck. The mixing tank was filled with approximately 2 m^3 of water and salt until a density of 1.33 g/cm^3 was reached. The throughput was set as 300-500 kg/h for solid material and 10 m^3/h for the liquid medium. This represents a solid liquid ratio of 30-50 g/L which was recommended by the machine provider. A centrifugal speed of 1800 rounds per minute was adjusted.

Process description

For the density separation, the big bag was fixed above the feed hopper and cut open. A conveyor belt transported the material into the mixing tank where it was stirred. From the middle of the mixing tank, the suspension was pumped into the centrifuge passing by a heavy metal trap which was not activated during the experiment.

In the centrifuge, radial forces caused the suspension to form a ring allowing air and dirt to be removed. The denser materials were pushed outwards whereas the lighter materials floated inwardly. A screw conveyor transported the materials to opposite ends lifting them out of the water surface (Figure 6.2). The dewatered solid fractions were then collected in big bags and the suspension was returned to the mixing tank. The output materials were visually assessed, and a reading of the power output was taken at full load.

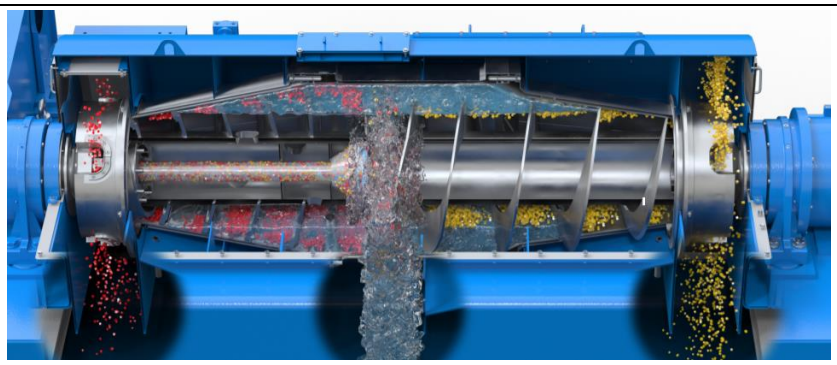


Image adapted from: Andritz Separation, n.d.

Figure 6.2: Centrifuge with denser (red) and lighter (yellow) material fractions

Technical challenges and process data

The initial big bag was too high to fit in between the roof of the truck and the feed hopper. Consequently, the material had to be transferred to a smaller big bag first.

After setting up the centrifuge the process worked smoothly. However, with increasing time, the material throughput reduced significantly. This was caused by rope material which clogged the pump. Several attempts to remove the DFG failed so that the experiment could not be finished. In the future, a lower solid liquid ratio should be adjusted to reduce the risk of blockages.

The measured power output is presented in Table 6.5.

Table 6.5: Power output from centrifugal density separation

Unit	Main Feed			Decanter			Total
	L1	L2	L3	L1	L2	L3	
kW	0.57	0.59	0.58	0.45	0	0	2.19

Output analysis

Most of the initially processed rope material entered the light fraction. This was expected as the rope material had a relatively low level of contamination. The rope material comprised of small fibres similarly to wool and were difficult to entangle.

6.1.3 Scale-up

Suitability for large-scale operations

The manual sink-float experiment at Vecoplan was inefficient and not suitable for a large scale-production. However, equipment for an automated sink-float separation exists and is used for example by Plastix Global to separate EOL fishing gear. Although DFG is much more contaminated and entangled than EOL fishing gear, the experiments showed that a sink-float

separation works in principle. Therefore, an automated sink-float separation can be assumed to be suitable for the large-scale treatment of DFG.

The experiment with centrifuges at Andritz Separation could not be completed, because the rope material caused blockages in the pump. To resolve this, a decreased solid liquid ratio was suggested. This would increase the water consumption or decrease the material throughput reducing the process efficiency. Although the reduction in the process efficiency could not be quantified, it is not expected to compromise the technology's suitability for a large-scale production. This is because, other companies such as Aquafil apply centrifuges to treat EOL fishing gear and carpets as well. Therefore, centrifuges are assumed to be suitable for the large-scale treatment of DFG.

Technology selection

Compared to sink-float techniques, centrifuges can be expected to result in a higher dirt removal because they apply stronger forces during the separation. As this is desirable for DFG, a centrifugal density separation was selected for the LCI model.

6.1.4 Life cycle inventory

Included activities

The density separation involves two process sequences of (1) charging, (2) suspending and mixing and (3) density separation (Figure 6.3). For the energy recovery scenario, only the first and for the mechanical recycling scenario both processing sequences are modelled (Figure 6.3).

The manual charging has been excluded from the modelling because no major environmental impacts were expected to result from this step. Also, the separation of lead from the bottom fraction was not considered. It was assumed that lead can be effectively separated through screens.

Included flows

For the density separation the following flows were included: (1) the DFG_3 input, the DFG_4 in- and output as well as the bottom fraction and its lead and residues output, and (2) the water, salt and electricity consumption (Figure 6.3).

Potential emissions to water were not included because the water cycle is designed as a closed loop. However, depending on the feedstock, the water quality may be reduced over time so that

a neutralisation or cleaning step may be required. Therefore, a wastewater treatment should be included in future LCIs.

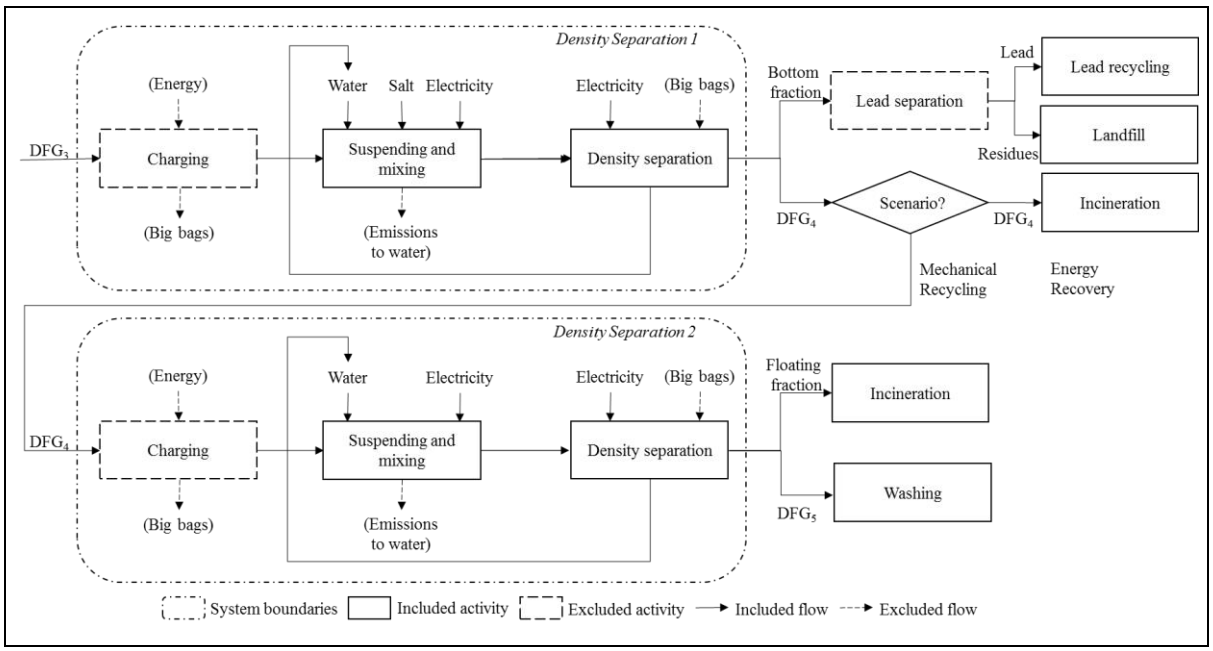


Figure 6.3: LCI scope for the density separation

In- and output flows

The DFG_3 input derives from the DFG_3 output from shredding (Table 5.14).

The lead output is directly taken from the DFG waste composition (Table 4.1) and thus assumes a separation efficiency of 100%. A complete material removal represents an idealistic scenario which is unlikely to be achieved. Still, during the sink-float experiment a separation efficiency of approximately 95% for lead was achieved (Table 6.6). As centrifuges are expected to yield higher separation efficiencies, a 5% higher separation efficiency for lead can be justified.

Table 6.6: Separation efficiency for lead

Lead content	Gillnets	Separation efficiency	Background data
	[%]	[%]	
In shredded DFG	2.74[a)]	100	Table 5.12
Removed during Density Separation 1	2.6	94.8	Table 6.4
Remaining after Density Separation 1	0.1	5.2	Calculated based on mass balance

[a)] Based on Schurig (2017a; Table 5.12). The up to 10 times higher values from gasification trials (Table 7.6) were not adopted because they represent an accumulated lead content that remained at the bottom of the big bag after the density separation.

The residues output is calculated by multiplying the mineral content in the DFG composition with its corresponding separation efficiency. As separation efficiency a value of 90% was assumed. This is a conservative estimate which lies below the claimed 99.9% in other applications (Andritz Separation, n.d.).

The bottom fraction and DFG_4 output results from mass balance calculations (Table 6.7).

Table 6.7: In- and output flows for Density Separation 1

Name	Mixed DFG	Unit	Trawl nets	Gillnets	Background data
Input					
DFG_3	909	kg	926	893	Table 5.14
Output					
Minerals	377	kg	507	248	Separation efficiency = 90%
Lead	67.6	kg	0	135	Separation efficiency = 100%
DFG_4	464	kg	419	510	Calculated based on mass balance

The DFG input for the second density separation derives from the DFG output from the first density separation (Table 6.7).

The floating fraction is determined by multiplying the PE/PP content in DFG with its separation efficiency. Based on separation efficiencies of up to 99.9% from similar applications (Andritz Separation, n.d.), a separation efficiency of 100% was assumed for PE/PP from DFG.

The remaining DFG output is calculated based on mass balance (Table 6.8).

Table 6.8: In- and output flows for Density Separation 2

Name	Mixed DFG	Unit	Trawl nets	Gillnets	Background data
Input					
DFG_4	464	kg	419	510	Results from Density Separation 1
Output					
Floating fraction	23.2	kg	0	46.3	Separation efficiency = 100%
DFG_5	441	kg	419	463	Calculated based on mass balance

Ancillary products and energy

The water consumption is established as 0.3 L/kg feed material (Table 6.9). This is based on the centrifuge's technical specifications for mixed polymers, stating a water requirement of 150L/h and an average material throughput of 500 kg/h (Andritz Separation, n.d.).

The salt requirement of 84.4 g/kg DFG is calculated based on the experimental sink-float separation and the above stated water consumption. It assumes that approximately 225 kg salt are needed to prepare an appropriate concentration in 800 L of water. Salt is only required for the first density separation (Figure 6.3).

The electricity consumption was calculated as 4.38 Wh/kg feed material based on the measured power output of 2.19 kW and a material throughput of 500 kg/h. However, compared to the installed 66kW in the centrifuge (Andritz Separation, n.d.) the measured power output seemed

very small. Thus, an average power output of approximately 34.1 kW was assumed to derive at an electricity consumption of 68.2 Wh/kg feed material (Table 6.9).

Table 6.9: Life cycle inventory for the density separation of 1 kg DFG_{3-4}

Name		Value	Unit	Probability Distribution				Background data
				T	Min		Max	
				Log	SD	BU	Pedigree Score	
Ancillary products								
	Water	0.3	L	Log	1.64	1.05	4,5,5,5,1	150L/h, for 1/500 h/kg
	Salt	84.4	g	Log	2.05	1.05	1,5,1,1,5	225 kg / 800 L
	Electricity	68.2	Wh	T	4.38		132	2.19-66 kW for 1/500 h/kg

6.1.5 Critical aspects

Separation efficiencies

In previous experiments for mixed plastic waste, separation efficiencies of 98% and 99.6% have been achieved with sink float and centrifugal techniques respectively (Shonfield, 2008). This supports the selection of centrifuges for a DFG treatment and the assumed separation efficiencies of 100% for lead and polymers. However, it also shows that the assumed 90% separation efficiency for the residues may be too low. Still, given the proximity of the density ranges between nylon and sediments (Table 6.1) and the high level of contamination and entanglement, the selected lower separation efficiency can be justified. Besides, the separation efficiency for the residues is not expected to significantly influence the outcome of the LCA or the quality of the recycled materials because the subsequent washing step is designed to remove the sediments that remain in the DFG output.

Energy consumption

The modelled energy consumption for the density separation was between 4.38 and 132 kWh/t which generally agrees with the literature. For example, Shonfield (2008) reported an average electricity consumption of 112 kWh/t for mixed plastics using a centrifugal density separation. Vecoplan designed a sink-float separator for agricultural film with a maximum power consumption of 74.4 kWh/t (U. Kramer, personal communication, 27 March 2019) and Turbo Laminare Trenntechnik (TLT) built a separator with an electricity consumption of approximately 15.1 kWh/t plastic (Shonfield, 2008). Although there are large variations, the literature values lie within the modelled uncertainty range so that no additional sensitivity analysis is conducted.

Output quality

The high chlorine content of 6.55% in the DFG_4 output would be problematic for incineration plants because it exceeds the generally accepted threshold of 4% (Interessengemeinschaft der thermischen Abfallbehandlungsanlagen in Deutschland [ITAD], n.d.). This means that an additional washing process would be required prior to incineration. It is assumed that a simple water rinsing can reduce the chlorine content to below 4%. This process was not modelled however, as it was not expected to result in in significant environmental impacts.

6.2 Washing

6.2.1 Experiment

Context and aim

Within MARELITT Baltic, a DFG washing experiment was conducted by Vecoplan at their test center in Bad Marienberg. This had the aim to remove sediments, salt and possibly other contaminants in preparation of a mechanical recycling.

Material and setup

All output material from the sink-float density separation was used for the washing experiment. This was approximately 36 kg of trawl nets ropes, 30.2 kg of clean trawl nets and 87.2 kg of dirty trawl nets as well as approximately 9.73 kg of rough sorted trawl nets, 213 kg of rough sorted gillnets and 45.2 kg of fine sorted gillnets (Table 6.4).

The experiments were carried out with the friction washer "Vecoplan HydroDyn" and its programme for agricultural foil. The material and water throughput were set as 450-600 kg/h and 15m^3/h respectively. As chemicals, 0.04 L of PolySepar CFX 1088 and 2-6g of PolySepar PK 1455 were added for the treatment of 1m^3 wastewater (U. Kramer, personal communication, 23 October 2017). Sodium hydroxide is added automatically, when a change of the pH-value was detected. For the handling a forklift, crane and several big bags were also employed.

In preparation of the washing, the fine sorted gillnets and the rough sorted trawl nets were used to flush out non-DFG material from previous experiments. As such, those materials are not considered further.

Process description

The DFG was emptied into a feed hopper (Figure 6.4) and a dosing screw transported it to a plastic cleaning unit [PCU]. Inside the PCU, the DFG was mixed with water and pressed through two opposite rotating discs with specific surface shapes (Figure 6.4). This resulted in high radial forces and friction which separated the DFG from the attached contaminants. The suspension was then pumped into the water separation unit [WSU].

Inside the WSU, a rotating drum pushed the suspensions outwards against a 2.5 mm screen. The larger materials remained in the drum and moved forward to be collected in big bags. The smaller particles and water were directed to a second screen with a 0.5 mm mesh size (Figure 6.4). Water and very fine particles passed through the screen and entered the wastewater treatment (Figure 6.4) whereas the larger materials were collected in a big bag (Figure 6.4).

The wastewater treatment consisted of a coagulation and a flocculation step. During coagulation the chemical PolySepar CFX 1088 destabilised the suspension so that the solid particles were freed. During flocculation, stirring and the addition of the chemical PolySepar PK 1455, agglomerated the solid particles which were then skimmed off and collected in a big bag.

Charging of the washing machine	Friction discs	0.5mm Screen	Wastewater

Figure 6.4: Visulisation of the washing process

Immediately after washing, the DFG output was visually assessed and weighted. Samples were taken and sent to EREMA and MAKSC for further analysis.

The residual screen fraction and the wastewater treatment residues were visually assessed, but contamination from previous experiments and time constraints prevented a meaningful weight measurement. Samples from the rough sorted gillnets were collected and sent to TU Clausthal for characterisation.

Technical challenges and process data

The washing worked well for the above-mentioned materials. The pH-value of the wastewater did not change during the experiments so that no sodium hydroxide was added.

Output analysis

The visual assessment of the DFG washing output revealed a notable reduction of the sediment content compared to the output from shredding (Figure 6.5).

The fine sorted trawl nets had a weight of 33 kg, 37 kg and 111 kg for ropes, clean nets and dirty nets respectively and the rough sorted gillnets weighed 54 kg.

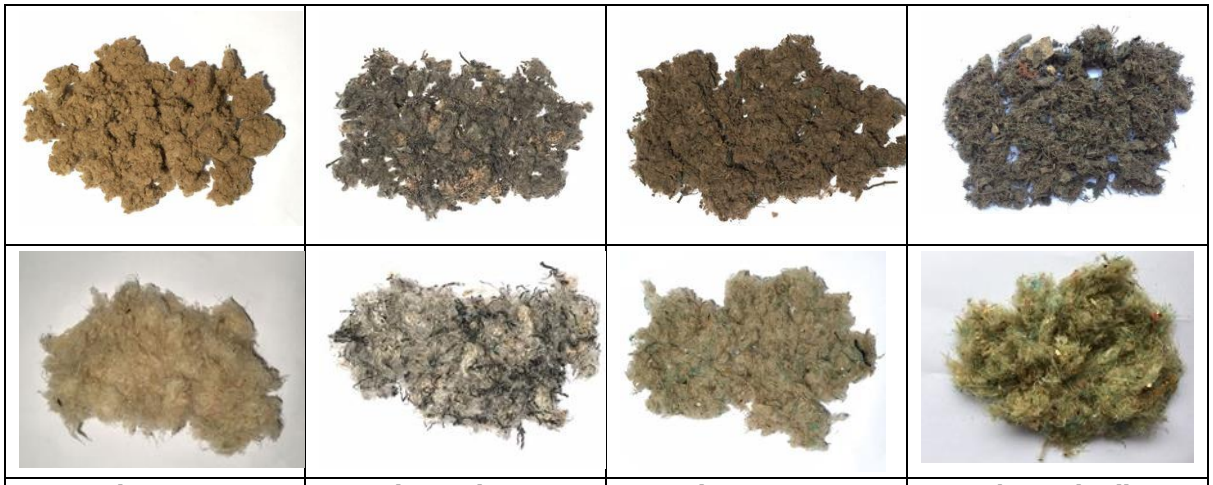


Figure 6.5: Comparison of DFG fractions after shredding (up) and washing (below)

The analysis of the DFG output is summarised in Table 6.10 and Table 6.11. As expected, nylon 6 was found as the main material. Other detected materials were PE, PP and PET in trawl nets and additionally polystyrene [PS], wood and aluminium [Al] in gillnets. The water content ranged between 14.9% - 31.2% and an ash content of 0.13% - 0.64% was measured.

Table 6.10: Material composition and contamination results

Analysis	Unit	Trawl nets, fine sorted			Gillnets, rough sorted	Reference
		Ropes	Clean	Dirty		
Main Material	-	-	PA6	PA6	PA6	Steiner, 2017
	-	PA6	PA6		PA6	Lehmann et al., 2018
Contamination	-	-	-	-	PE, PP, PS, wood, Al	Steiner, 2017
	-	PP, PET	PE, PP, PET		PE, PP, PET, wood	Lehmann et al., 2018
Water content	%	-	17.52	14.93	31.19	Steiner, 2017
	%	>5.0	>5.0		>5.0	Lehmann et al., 2018
Ash content	%	-	0.13	0.50	0.64	Steiner, 2017

All washed output materials showed elevated chlorine and bromine values. High values for lead were only detected in gillnet material. Other relevant chemical substances were analysed

as well (Lehmann et al., 2018). However, they did not show elevated values and were thus not included here.

Table 6.11: Heavy metal concentration in washing output

Output	F [ppm]	Cl [ppm]	Br [ppm]	Cd [ppm]	Cr [ppm]	Hg [ppm]	Pb [ppm]	Sb [ppm]
Trawl nets, ropes	<50	973	<193.5	0.72	<10	<0.1	10.1	<10
Trawl nets, clean and dirty	<50	1421	<159.7	7.20	<10	<0.1	25.7	<10
Gillnets, rough sorted	<50	1011	<96.8	0.30	<10	<0.1	358	28.2

Table adapted from: Lehmann et al., 2018

The visual examination of the residues exposed predominantly polymer fibres in the screen fraction (Figure 6.6) and sediments in the wastewater treatment residues.

0.5 mm Screen

Trawl nets Ropes

Rough sorted Gillnets

Figure 6.6: Residual screen fractions from the washing of DFG

The results of the external analysis of the washing residues for rough sorted gillnets are provided in Table 6.12.

Table 6.12: Analysis of the residual washing output for rough sorted gillnet material

Sample	As [ppm]	Pb [ppm]	Cd [ppm]	Cr [ppm]	Cu [ppm]	Ni [ppm]	Co [ppm]	Mn [ppm]	Fe [ppm]	Zn [ppm]
Screen	1.2	539	0.3	5	36	9	22	80	1216	85
Residues	0.6	85	1.2	21	58	10	2.9	44	2424	78

Table adapted from: Sommer and Hoffmann, 2017

Waste composition

An overview of the washing in- and output flows is provided in Table 6.13. The DFG input weights were directly based on the density separation (Table 6.4), because all material was processed. As such they reflect the previously assumed water content of 3%. The DFG output was assumed to contain a water content of 35%. The resulting water uptake was calculated based on the mass balance and accounted for as an additional input flow (Table 6.13).

Based on visual observations, the residual screen fraction was estimated as 15% for trawl nets and as 50% for gillnets. The difference can be explained by a higher amount of monofilament fibres in gillnets compared to trawl nets. The residues fraction was calculated with an assumed

separation efficiency of 98% for sediments (U. Kramer, personal communication, 05 April 2019) and the measured ash contents (Table 6.10). For trawl net ropes an average ash content of 0.315% was assumed. The material loss was calculated as the difference between the in- and output flows (Table 6.13). The described approach resulted in negative values for the material loss of clean and dirty trawl nets (Table 6.13). A material gain may be explained by the initially present material stock inside the washing machine. In fact, a complete removal of previously processed materials was not possible as this would have required the machine's disassembly.

Table 6.13: Overview of DFG composition

Process	Flow	Unit	Trawl nets				Gillnets
			Fine sorted			Total	Rough sorted
			Ropes	Clean	Dirty		
Washing	Input						
	DFG_5	kg	36.0 [a)]	30.2 [a)]	87.1 [a)]	153	213 [a)]
		%	100	100	100	100	100
	Water	kg	10.9 [b)]	12.2 [b)]	36.6 [b)]	59.7	17.8 [b)]
		%	30.2	40.4	42.0	38.9	8.4
	Output						
	Screen (PA)	kg	5.40 [c)]	4.53 [c)]	13.1 [c)]	23.0	106 [c)]
		%	15.0	15.0	15.0	**15.0**	**50.0**
	Residues (Minerals)	kg	5.25 [d)]	2.43 [d)]	28.0 [d)]	35.7	17.5 [d)]
		%	14.6	8.0	32.2	**23.3**	**8.2**
	Material Loss (% comp.)	kg	3.24 [b)]	-1.56 [b)]	-28.4 [b)]	-26.7	52.6 [b)]
		%	9.0	-5.2	-32.6	**-17.4**	**24.8**
	DFG_6 + Water uptake	kg	33	37	111	181	54
		%	91.7	122.5	127.4	118.1	25.4
Extrapolated Laboratory analysis	Input						
	DFG_6 + Water uptake	kg	33	37	111	181	54
		%	100	100	100	100	100
	Output						
	Ash content	kg	0.104 [e)]	0.048	0.555	0.707	0.346
		%	0.315	0.13	0.5	0.391	0.64
	Water content	kg	11.6	13.0	38.9	63.4	18.9
		%	35	35	35	35	35
Remaining Material	Input						
	DFG_6	kg	22.1	24.8	74.7	121	36.2
		%	100	100	100	100	100
	Output						
	Water (%Min. + %PA)	kg	0.663	0.744	2.23	3.64	1.09
		%	3	3	3	**3**	**3**
	Ash (Minerals)	kg	0.104 [f)]	0.048 [f)]	0.555 [f)]	0.707	0.346 [f)]
		%	0.5	0.2	0.7	**0.6**	**1.0**
	Rest (Polymers-PA)	kg	21.3 [b)]	24.0 [b)]	71.6 [b)]	117	34.8 [b)]
		%	96.5	96.8	96.3	**96.4**	**96.0**

Factor: (98:2) / 0.97

"Water content" - "Water"

[a)] The value is based on Table 6.4. [b)] The value is calculated based on mass balance. [c)] The value is estimated based on visual observations. [d)] The value is extrapolated from the ash content and an assumed separation efficiency of 98%. [e)] The value assumes an average ash content for trawl nets. [f)] The value is based on the laboratory analysis.

To establish the dry DFG composition (Table 4.1), the screen fraction was allocated to the nylon and the residues were allocated to the minerals content (Appendix A). The material loss was proportionally allocated across different waste categories to reflect the composition of the DFG input.

The remaining materials in the DFG output were divided into a water, ash and rest fraction (Table 6.13). To establish the dry DFG composition, the ash ratio was allocated to the mineral and the rest ratio to the nylon fraction (Appendix A).

6.2.2 Scale-up

Suitability for large scale operation

The use of the Vecoplan Hydrodyn washing machine resulted in a notable quality improvement. However, the abrasive material content was too high and would have caused excessive wear during an extrusion (R. Steiner, personal communication, 02 August 2017). Therefore, a better pre-cleaning for example as part of a centrifugal density separation or a second washing step is required. Furthermore, the DFG output was relatively low. To increase the nylon yield, a smaller screen size should be applied in the water separation unit. With those modifications the suitability for a large-scale operation is assumed.

Technology selection

For the LCI model, the Vecoplan Hydrodyn washing machine is selected because it proved suitable to treat DFG at a larger scale. In the future, dry cleaning techniques should be also considered due to their expected lower environmental impact. For a large-scale operation no material loss is assumed.

6.2.3 Life cycle inventory

Included activities

The washing includes the (1) charging, (2) friction washing and (3) water separation as well as the (4) screening and (5) waste water treatment (Figure 6.7). For the LCI the charging and the screening were not considered due to their expected low environmental impact.

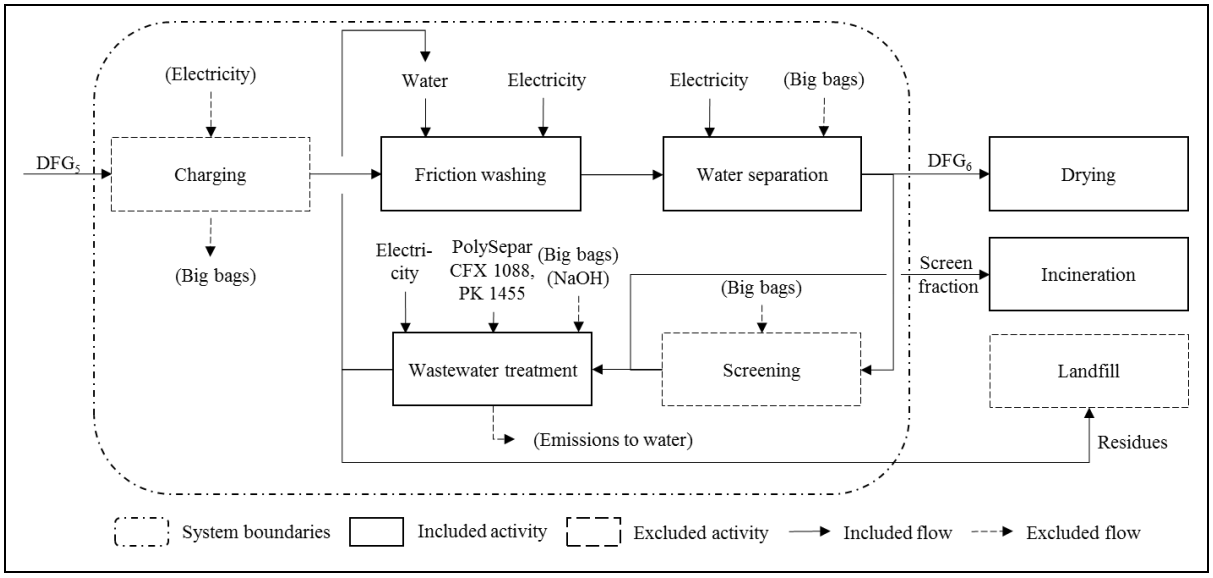


Figure 6.7: LCI scope for the washing

Included flows

The modelled in- and output flows for the washing are the (1) DFG_5 input, DFG_6 output and the screen fraction as well as the residues output and the (2) electricity, water and chemicals consumption. The production of big bags and sodium hydroxide was excluded. Similarly, the emissions to water were not considered because of the closed water cycle (Figure 6.7).

In- and output flows

The DFG_5 input equals the DFG_5 output from the second density separation (Table 6.14). The screen fraction is calculated based on the nylon content in DFG and its relevant separation efficiency. Based on similar applications with an optimised screen size, a separation efficiency of 15% was assumed (U. Kramer, personal communication, 05 April 2019; Table 6.14).

The output residues are directly based on the remaining mineral content of the DFG. It therefore assumes a separation efficiency of 100% which overestimates the technology's capability. However, typical separation efficiencies for the washing lie around 98% (U. Kramer, personal communication, 05 April 2019) so that a nearly complete removal can be expected. The remaining DFG_6 material is calculated based on the mass balance (Table 6.14).

Table 6.14: In- and output flows for Washing

Name	Mixed DFG	Unit	Trawl nets	Gillnets	Background data
Input					
DFG_5	441	kg	419	463	Results from Density Separation 2
Output					
Screen	59.8	kg	54.3	65.4	Separation efficiency = 15%
Residues	41.9	kg	56.3	27.5	Separation efficiency = 100%
DFG_6	339	kg	308	370	Calculated based on mass balance

Ancillary products and energy

The total electricity consumption of the washing process was estimated as 100 kWh/t DFG (U. Kramer, personal communication, 23 October 2017).

For the water loss and corresponding water consumption, values of 1-3m3/t were predicted for DFG (U. Kramer, personal communication, 23 October 2017).

Based on the experimental setup (Chapter 6.2.1), the consumption of PolySepar CFX 1088 and PolySepar PK 1455 was calculated as 1 ml/kg DFG and as 0.05-0.15 g/kg DFG respectively.

Table 6.15: Life cycle inventory for the washing of 1 kg DFG

Name	Value	Unit	Probability Distribution				Background data
			T	Min		Max	
			Log	SD	BU	Pedigree Score	
Ancillary products							
PolySepar CFX 1088 [a)]	1	mL	Log	1.24	1.05	4,4,1,1,1	0.04 L/m^3, 15 m^3/h, 600 kg/h
PolySepar PK 1455 [b)]	0.1	g	T	0.05		0.15	2-6 g/m^3,15 m^3/h, 600 kg/h
Water	2	L	T	1		3	1-3 m^3/t
Electricity	100	Wh	Log	1.62	1.05	4,5,1,1,4	100 kWh / 1000 kg

[a)] Aluminium hydroxide was modelled as a proxy for PolySepar CFX 1088. For the unit conversion a density of 2.42 L/kg was assumed. [b)] Cationic resign was used as a proxy for PolySepar PK 1455.

6.2.4 Critical aspects

Screen output

The screen fraction represents an undesired polymer loss which should be reduced in the future. The washing experiments showed a relatively large screen fraction of up to 50% due to the use of an inadequate screen size. Based on the machine provider, a reduced polymer loss of 15% can be expected for large scale operations. This is in line with other studies on mixed plastic waste which determined a polymer loss of approximately 10% (Shonfield, 2008). Therefore, the modelled separation efficiency of 15% appears a good estimate.

Water consumption

The Vecoplan HydroDyn process has a water throughput of 25 - 33.3 m^3/t. This is approximately ten times higher than the benchmark processes which run at 2-3m^3/t (Hopewell, Dvorak and Kosior, 2009). However, the Vecoplan HydroDyn process operates a closed water cycle which provides for most of its own water needs. This means that water is only added to replace the water losses.

The water losses were estimated at 1-3 m^3/t for DFG. If all lost water is added to the output fractions, a water uptake of 100-300% would be expected. In this case, the water content in the output fractions would account for at least 50-75% which is higher than the estimated 35%. The difference can be explained by water leakages from the big bags in which the output materials were collected.

Electricity consumption

The electricity consumption of the Vecoplan HydroDyn process was estimated at 100 kW/t. This is roughly four times higher than the Pla.To dry cleaning process which has a maximum power output of 90-110 kW at a material throughput of 4 t/h for mixed plastic waste (Shonfield, 2008). Therefore, dry cleaning techniques should be considered in the future.

6.3 Drying

6.3.1 Experiments

Context and aim

Within MARELITT Baltic, the Magdeburger Kunststoff-Service-Center [MAKSC] and the Hochschule Magdeburg conducted small-scale drying experiments for DFG. This had the aim to prepare the material for a mechanical recycling and to avoid mould and odour formation.

Materials and setup

MAKSC was provided with washed fibres from rough sorted gillnets and trawl nets ropes at approximately 5 kg each. Two smaller samples of clean and dirty trawl nets were joined to a 5 kg sample as well. The Hochschule Magdeburg received the remaining washed fibres from rough sorted gillnets which accounted for approximately 50 kg.

Process description

The drying took place in laboratory heating cabinets with recirculating air. At the Hochschule Magdeburg DFG was dried until no further weight loss could be observed while MAKSC dried DFG for 4 hours at 80°C (J. Radek, personal communication, 22 November 2018).

Output analysis

MAKSC reported a water content below 0.2% for the output DFG. This was also assumed for the fibres at the Hochschule Magdeburg.

6.3.2 Scale-up

Suitability for large scale operation

The laboratory drying worked well but is not suitable to treat large quantities of DFG. Therefore, suitable industrial drying processes need to be sought.

At Plastix Global a convective drying process is used to process EOL fishing gear (H.A. Kristensen, personal communication, 21 November 2017). While data for this dryer could not be obtained, Andritz Separation provided information for a similar machine. The large-scale suitability of this dryer was assumed based on expectations of the machine producer but without confirming it in experiments with DFG.

Technology selection

As technology for the LCI model, a conductive drying process from Andritz Separation was selected due to its data availability and expected large-scale applicability.

Technology description

The modelled paddle dryer is designed for wet material with a water content of approximately 35%. As parameters, a material throughput of 680 kg/h, a steam consumption of 337 kg/h at 165°C and a material output with a water content of roughly 2.6% can be expected.

The drying process can be divided into a charging, drying and optional energy recovery (Figure 6.8). During the charging the big bag is removed, and the feedstock is loaded into the machine. As part of the drying, heated screws transport the feedstock through the heating chamber. The contained water evaporates and is either released to the atmosphere or condensed and used for a district heating. The dried output leaves the heating chamber and is collected in big bags. The saturated steam which is used to heat the screws can be reheated and returned to the process.

6.3.3 Life cycle inventory

Included activities

The modelled drying process is depicted in Figure 6.8. The charging was not expected to significantly contribute to the environmental impacts and was thus excluded. It was assumed that the evaporated water was directly released to the atmosphere (Figure 6.8).

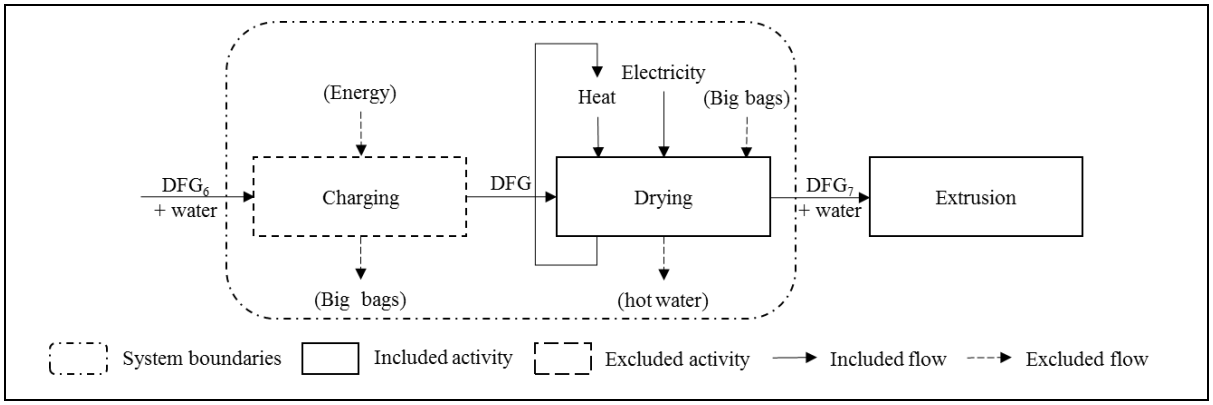


Figure 6.8: LCI scope for the drying

Included flows

The following in- and output flows were considered: the (1) DFG_6 input and the DFG_7 output as well as the (2) electricity and heat consumption (Figure 6.8). The emissions to air were not considered because no harmful substance was expected to evaporate during the process.

In- and output flows

The DFG input is directly based on the DFG output from the washing (Table 6.14). As no DFG material is removed during the drying, the DFG output equals the DFG input (Table 6.16).

Table 6.16: DFG in- and output for drying

Name	Mixed DFG	Unit	Trawl nets	Gillnets	Background data
Input					
DFG_6	339	kg	308	370	Results from Washing - Table 6.14
Output					
DFG_7	339	kg	308	370	No DFG material (only water) is removed.

Ancillary products and energy

The electricity consumption is calculated as 28.4 Wh/kg (Table 6.17), assuming a material throughput of 680 kg/h and an average power output of 19.3 kW (M. Maingay, pers. comm. 02 April 2019)

The heat consumption is estimated at 10 times the electricity consumption (M. Maingay, pers. comm. 02 April 2019) which corresponds to approximately 284 Wh/kg (Table 6.17).

Table 6.17: Life cycle inventory for the drying of 1 kg input material

Name	Value	Unit	Log	SD	BU	Pedigree Score	Background data
Ancillary products and energy							
Electricity	28.4	Wh	Log	1.89	1.05	4,5,5,5,4	680 kg/h, 19.3 kW
Heat	284	Wh	Log	1.89	1.05	4,5,5,5,4	284 kWh/t input

6.3.4 Critical aspects

Energy consumption

At a material throughput of 1.25 kg/h (Chapter 6.3.1) and an average power output of 850 W (Binder GmbH, 2018), a laboratory drying process consumes approximately 680 kWh/t DFG. This is more than twice of the modelled drying process. However, this was expected as the laboratory dryer is not designed for a large-scale operation. Therefore, the modelled drying process must be compared with other industrial machines. For example, Vecoplan designed a mechanical dryer with a throughput of 1 t/h for agricultural film, comprising of a 160 kW centrifuge, a 120 kW air heating unit and a 22 kW feed blower (U. Kramer, personal communication 27 March 2019). It results in an electricity consumption of 302 kWh/t which is comparable to the energy consumption of 312 kWh of the modelled thermal drying technique.

7 Extrusion and Gasification

7.1 Extrusion

7.1.1 Experiments

Context and aim

As part of MARELITT Baltic, the Hochschule Magdeburg carried out an extrusion experiment with DFG to investigate its general suitability for a mechanical recycling. Furthermore, the Magdeburger Kunststoff-Service-Center [MAKSC] extruded DFG material to determine its specific mechanical properties.

Materials and setup

For the extrusion, the Hochschule Magdeburg and the MAKSC used a fraction of their previously dried DFG (Chapter 6.3.1). Both institutes processed rough sorted gillnets while MAKSC also processed trawl nets ropes and a mixed fraction of clean and dirty trawl nets.

Process description

At the Hochschule Magdeburg, a laboratory ram extruder was equipped with a 1 mm square shaped die and loaded with the material. The barrel and thus the material were preheated to 220ºC, 230ºC and 240ºC in three consecutive trials before a pressure of 600 kN was applied for 10 minutes (Weißbach and Gerke, 2018). After a cooling period of approximately 15 min the plates were removed at roughly 30ºC (Ibid). The output plates were then visually assessed.

At MAKSC, a laboratory single screw extruder was equipped with an 800 µm filter (Lehmann et al., 2018). The barrel was preheated to 266-274 ºC before the material was charged into the machine. The rotating screws and a pressure of up to 75 bar pressed the material through the filter and a die (Ibid). The three separate strands coming out of the die were cooled down in a water bath (Figure 7.1) before a strand cutter chopped them into pellets. The filter residues and pellets were visually assessed. The pellets were then moulded into test specimen and their mechanical properties were determined through MAKSC's laboratory.

Figure 7.1: Visualisation of the screw extrusion experiments at MAKSC

Technical challenges

During the ram extrusion, no technical complications were reported. The screw extrusion was not immediately possible because the low pourability and bulk density of DFG fibres did not allow an automated loading. The material had to be inserted manually to reach the required compaction (Lehmann et al., 2018).

Output analysis

The ram-extruded plates at the Hochschule Magdeburg revealed a smooth surface area with a random colour pattern (Figure 7.2). The black material in the plates (Figure 7.2) was identified as rubber contamination (Weißbach and Gerke, 2018). Near the rubber small holes and thus potential breaking points were detected.

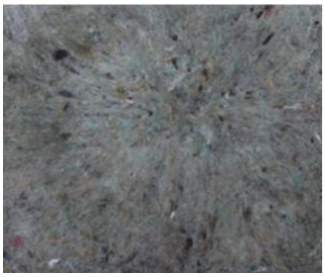
		
Input fibres from DFG	Output plates	Rubber contamination
Images from: Weißbach and Gerke, 2018		

Figure 7.2: In- and output of ram extrusion experiments at the Hochschule Magdeburg

The filter residues from the screw extrusion at MAKSC (Figure 7.3) revealed up to 2 mm long metal residues. The pellets had a homogenous black, brown and grey colour (Figure 7.3) and pellets from rough sorted gillnets had a typical wood smell (Lehmann et al., 2018).

		
		
Clean and dirty trawl nets	Trawl nets ropes	Rough sorted gillnets
Images from: Lehmann et al., 2018		

Figure 7.3: Output pellets and filter rest from screw extrusion experiments at MAKSC

The determined mechanical properties are summarised in the Table 7.1.

Table 7.1: Mechanical properties of injection moulded test specimen from DFG

Test	Property	Unit	Clean and dirty Trawl nets [a)]	Trawl nets Ropes [a)]	Rough sorted Gillnets [a)]	Virgin [b)]
Tensile Strength, @ maximum	Strength	MPa	56.3	52.7	27.1	90
Elongation, @ maximum	Ductility	%	3.7	3.3	1.5	4.5
Young's modulus	Stiffness	MPa	2635	2630	2401	3000
Impact strength	Toughness	kJ/m2	5.0	3.0	2.8	9

[a)] Adapted from: Lehmann et al., 2018; [b)] Adapted from Matweb, n.d.

7.1.2 Scale-up

Suitability for large scale operation

Ram extruders – as used by the HS Magdeburg – have a high product flexibility and minimal wear, but also a long heating time (Wagner, Mount and Giles, 2013) which makes them unfit for a large-scale operation.

Single screw extruders heat up much faster, but they also required a laborious manual loading. To solve this, twin extruders and feeding methods using pressurised air and vibrating trays were previously proposed (Dagli et al., 1995). Today, specific feeding units for an automated compaction of fibrous materials are available (EREMA, 2018; EREMA Recycling News, 2014). Therefore, single screw extruders are expected to be suitable for a large-scale operation.

Technology selection

Plastix Global and Bureo extrude EOL fishing gear into pellets on a large-scale. Both companies use single screw extruders from EREMA. Bureo uses an Intarema TE 1310 to process nylon fibres from EOL fishing gear. As, this is comparable to the nylon fibres from DFG, the same technology is used for the LCI model.

Process description

Operating at 280°C for nylon, the Intarema TE 1310 has a throughput of 500-550 kg/h and is typically equipped with a mesh filter of 100-130 µm (R. Binder, personal communication, 29 March 2019).

The extrusion entails a preparation, extrusion and strand cutting as well as a screening, drying and water cooling (). During the preparation, the material is loaded onto a conveyor belt, directed through a metal detector, mixed, cut and dried (EREMA, 2018).

During the extrusion the material is drawn in by the screw, melted, homogenised and filtered (EREMA, 2018). The materials with a low boiling point including volatile organic compounds and additives are evaporated, collected and condensed under vacuum to be disposed of in a hazardous landfill. The collected filter residue is sent for incineration or crushed and returned to the process. Master-batches can be added when specific colours and properties are required.

During the strand cutting, the melt strands are cooled down in water filled chutes and guided into the pelletiser for cutting (EREMA, 2018). The outcoming pellets are separated from the water at a screen. The pellets are then dried in a centrifuge and collected while the residual water is cooled down and returned to process.

7.1.3 Life cycle inventory

Included activities

The individual extrusion process steps are depicted in Figure 7.4. No process is excluded.

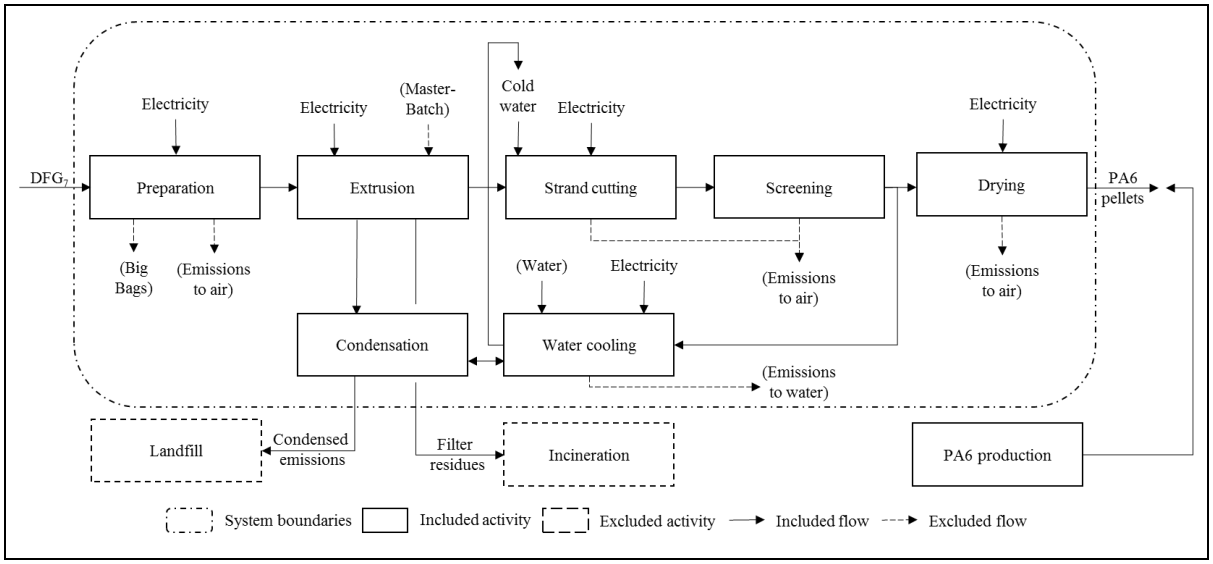


Figure 7.4: LCI scope for the extrusion

Included flows

The modelled in- and output flows are the (1) DFG_7 input, the condensed emissions and the filter residues as well as the pellets and the (2) electricity consumption.

The water consumption was described as "negligible low" (R. Binder, personal communication, 29 March 2019) and was thus not included. However, given that the cooling water is replaced on a weekly basis (Ibid), the water consumption and possible emissions to water should be considered in future LCIs. Emissions to air will be mainly in form of water, which was not included here. A master batch was not used for the experiments and thus not considered.

In- and output flows

The DFG input is directly based on the DFG output from drying (Table 6.16). The filter residue was estimated as 1-2.5% of the input material (R. Binder, 2019, personal communication, 29 March) and for the model an average separation efficiency of 1.75% was assumed. Similarly, the condensed emissions were estimated at 2-10 kg/day (Ibid). At a throughput of 12-13.2 t/day, this represents 0.2-0.8% so that an average separation efficiency of 0.05% was assumed. The remaining output pellets result from the mass balance (Table 7.2).

Table 7.2: In- and output flows for Extrusion

Name	Mixed DFG	Unit	Trawl nets	Gillnets	Background data
Input					
DFG_7	339	kg	308	370	Results from Drying, Table 6.16
Output					
Filter residues	5.94	kg	5.39	6.48	Nylon separation efficiency = 1.75%
Condensed emissions	0.170	kg	0.154	0.185	Nylon separation efficiency = 0.05%
Pellets	333	kg	302	364	Calculated based on mass balance

Avoided production

The produced pellets are assumed to replace virgin nylon pellets at a ratio of 1:1. However, this does not consider quality differences which are further discussed below.

Ancillary products and energy

The electricity consumption was estimated at 0.32-0.34 kWh/kg (R. Binder, personal communication, 29 March 2019). The water cooling requires an additional 10-15% (Ibid).

Table 7.3: Life cycle inventory for the extrusion of 1kg input material

Name	Value	Unit	Min	Max	Background data
Avoided production					
Nylon pellets	0.982	kg	0.974	0.990	1-2.5% Filter residues, 0.02-0.08% condensed emissions
Ancillary products					
Electricity	0.372	kWh	0.352	0.391	0.32-0.34 kWh/kg, + 10-15%

7.1.4 Critical aspects

Pellets output quantity

The pellets output was modelled as 98.2% and the joint material loss as 1.8%. This is in line with other experiments on mixed plastic waste which found a total material loss of 2% (Shonfield, 2008). Also, M. Lehmann who conducted the experiments at the MAKSC estimated a

material loss at 1% for DFG fibres (personal communication, 25 October 2018). Given the small variations in the reported values, a sensitivity analysis on this is not conducted.

Pellets output quality

The recycled pellets were modelled to replace virgin pellets at a ratio of 1:1. This represents a best-case which is commonly assumed in other LCAs (Shonfield, 2008; Wäger and Hischier, 2015; Beigbeder et al., 2019). However, there can be quality differences between recycled and virgin polymers, and it needs to be discussed whether this is also expected for DFG pellets.

A first indication of quality differences is the level of contamination. While contamination would not be expected in virgin material, DFG pellets did not only contain rubber and metal pieces (Figure 7.2) but also heavy metals and possibly other hazardous substances. For example, the lead content of 358 ppm in washed fibres from gillnets (Table 6.11) exceeds the maximum concentration limit of 100 ppm for lead, cadmium, mercury and chromium in packing materials (Directive 94/62/EC, 1994). Although the extrusion will have further reduced the lead content, the use of DFG pellets for packaging materials may not be allowed.

Another indication for a quality difference comes from the mechanical properties. When recycled DFG pellets are compared with Ultramid® B35 – a virgin nylon used for fishing nets – (Matweb, n.d.) the strength, ductility, stiffness and toughness (Table 7.1) decreased by 37%, 18%, 12% and 44% respectively. Even when compared with pellets from EOL fishing gear such as Longships (Fishy Filaments, 2018), only the tensile strength remained competitive. Although the mechanical properties may be improved with higher separation efficiencies at a larger scale, it would be unlikely to reach virgin like properties. In other words, significant quality differences can be expected.

To account for the inferior properties of recycled materials, a reduced market substitution factor can be adopted. In other LCAs, a substitution factor of 81% is typically selected for mixed plastics (Cremiato et al., 2018; Unger et al., 2017; Tunesi, Baroni and Boarini, 2016) – a value that was assumed for the sensitivity analysis in Chapter 10 as well.

Electricity consumption

The electricity consumption was modelled at 0.352-0.391 kWh/kg. This is comparable with other single screw extruders which consume 0.347-0.417 kWh/kg of electricity at temperatures of 220°C (Abeykoon et al., 2014). Similarly, Rigamonti et al. (2014) reported an electricity consumption of 0.24-0.47 kWh/kg and Shonfield (2008) reported slightly lower values of 0.24-0.30 kWh/kg presumably for polymers with a lower melting point.

7.2 Gasification

7.2.1 Experiment

Context and aim

Within MARELITT Baltic, a gasification experiment was conducted by Clean Carbon Conversion in Freienbach, Switzerland. This had the aim to investigate the suitability of their steam gasification process to produce valuable output materials from DFG.

Materials and setup

As feedstock, 312 kg of rough sorted, shredded gillnets with a water content of 25% was used. It was divided into two roughly equal fractions which were processed at different dates.

The experiment was conducted at a pilot plant with a throughput of 5 t/day. The setup and plant is depicted in Figure 7.5. It includes two consecutive silos, a rotary kiln reactor and a residue collection bin as well as a two-stage gas cleaning unit. The illustrated syngas collection was replaced with a combustion unit. The material throughput was set at 110 kg/h.

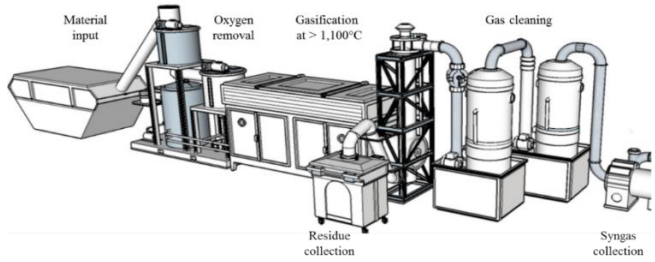


Image adapted from: Clean Carbon Conversion, n.d.

Figure 7.5: Steam gasification process (left: schemata; right: actual plant)

Process description

The DFG was charged into the first silo and flushed with nitrogen to prevent its combustion at high temperatures. In the second silo, the oxygen content was measured and if needed further reduced. Within the gasification reactor, temperatures above 1100°C let the materials with a lower boiling point such as hydrocarbons, halogens and several heavy metals enter the gas phase. The remaining materials were collected in a tightly closed metal bin at the end of the reactor. The gas was water quenched to below 70°C to avoid the formation of tar, dioxins and furans. It was then directed through an acid scrubber to remove heavy metals and through a basic scrubber to remove halogens. The formed solid materials in the scrubbers were recovered by a filter press and the cleaned gas was combusted.

Throughout the process the power output and the syngas composition were measured and recorded. After a cooling period the solid residues from the metal bin were collected, weighed and visually assessed.

Process data and technical challenges

The DFG blocked the auger several times during the experiment. Although, this could be fixed by temporally reversing the auger, it suggests that the feedstock was too large. Indeed, the pilot plant is designed for particles smaller than 25 mm, but some of the fine shredded fibres had approximately double that length. Therefore, a smaller screen size may be needed for shredding.

The recorded power output was used to determine the electricity consumption (Table 7.4). The data reflects an operation at full load and excludes the warming up and shut-down phase.

Table 7.4: Energy consumption of the steam gasification

Date	Duration [min]	Average power output [kW]	Electricity consumption [kWh]
05.12.2017	86	116	166
13.12.2017	93	143	221
Total	179	129	384

Output analysis

The solid residues (Figure 7.6) accounted for 108.5 kg and 42.5 kg during the first and second experiment respectively. The visual examination revealed a grey/black residue containing approximately 50% of lead fragments (F. Rupert, personal communication, 18 March 2019).


Images from: Clean Carbon Conversion, 2017

Figure 7.6: Solid residues from steam gasification

Table 7.5 shows the average composition of the output gas. During both experiments, hydrogen formed the largest gas fraction, followed by carbon monoxide, carbon dioxide and methane.

Table 7.5: DFG steam gasification gas composition

Date	Unit	H_2	CO	CO_2	CH_4
05/12/2017	%	48.5	24.9	12.1	9.2
13/12.2017	%	42.9	22.7	7.0	4.8

Waste composition

The in- and output flows for both experiments are summarised in Table 7.6. To allow conclusions on the dry waste composition, an assumed big bag weight of 1 kg and the water content of 25% were subtracted from the DFG input. To account for misplaced material during handling, a material loss of 1% was assumed. The solid residues were estimated to contain of equal amounts of lead and residues and the syngas output was calculated based on the mass balance.

Table 7.6: Waste composition from steam gasification of rough sorted gillnets

Process	Flow			[kg]	[%]
Gasification	Input				
		DFG + water		311[a)]	(100)
			Water	77.8	(25)
			DFG	233	100
	Output				
		Material loss (% input composition)		2.33	**1.0**
		Solid residues		151	64.8
			Lead (Lead)	75.5	**32.4**
			Residues (Minerals)	75.5	**32.4**
		Syngas (%PE/PP, %PA)		79.9 [b)]	**34.3**

[a)] The value excludes an assumed big bag weight of 1 kg. [b)] The value is based on the mass balance.

The lead content was attributed to the lead content in DFG. The residues were attributed to the mineral content and the syngas output was proportionally attributed to the polymer fractions. The material loss was allocated across different waste fractions to reflect the waste composition of the DFG input (Appendix A).

7.2.2 Scale-up

Suitability for large scale operation

The experiment has shown to successfully turn DFG into a hydrogen rich fuel gas. As such it was deemed suitable for a large-scale production.

Technology selection

Different air and steam gasification processes exist. For the LCI model the process from clean carbon conversion is selected because it proved suitable to treat DFG at a large scale. The experiments were conducted with a pilot plant. However, industrial plants are typically equipped with a cyclone to collect particulate matter prior to the wet scrubbing. This is also illustrated in Figure 7.7 below. Furthermore, during the experiment a water content of 25% was present in the DFG material. If water is not present, it needs to be added to the process.

7.2.3 Life cycle inventory

Included activities

The steam gasification process entails a preparation, gasification and residue treatment as well as a cyclone and wet scrubbing (Figure 7.7). The residue treatment is not considered but it is assumed that a low energy consuming screening process can separate the metallic lead from other residues. While the lead is recycled and the produced syngas is modelled to replace an average syngas production, the landfilling of residues was not included here.

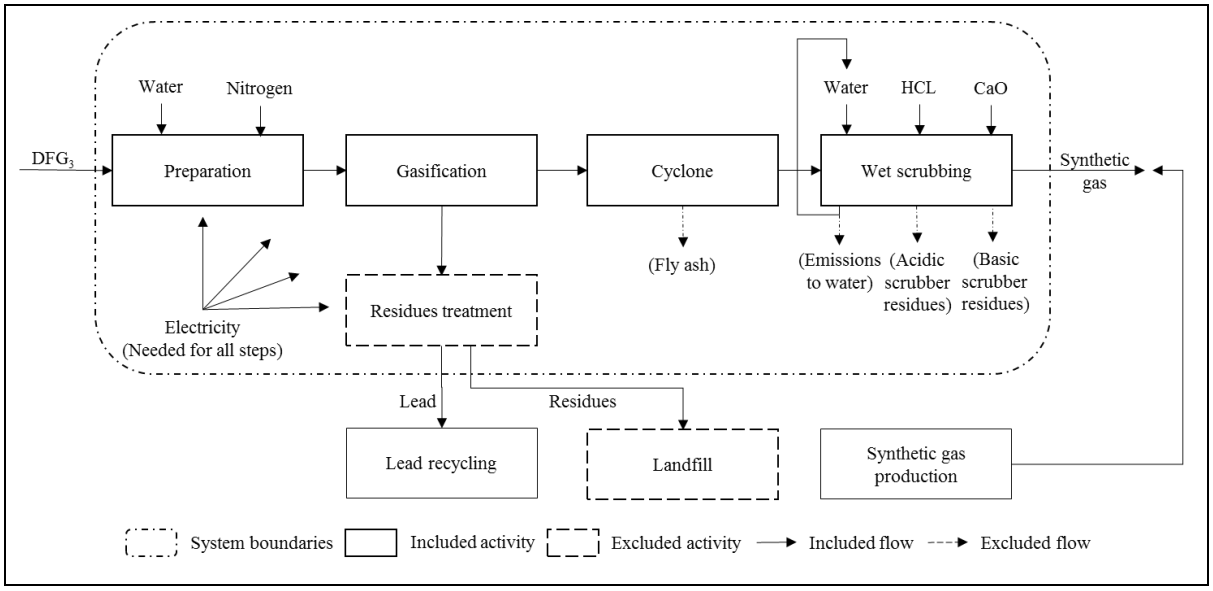


Figure 7.7: LCI scope for the steam gasification

Included flows

The considered in- and output flows are the (1) DFG_3 input, lead, residues and syngas output and the (2) water, nitrogen and energy as well as lime and hydrochloric acid consumption. The emissions from the closed loop water cycle are not modelled, but because wastewater is typically replaced every 3 months, emissions to water should be considered in the future.

In- and output flows

The DFG input is based on the DFG output from shredding (Table 5.14). The output flows are calculated by multiplying the waste content in DFG with their respective separation efficiencies. For lead and minerals, a separation efficiency of 100% was assumed. In the DFG input, a sulphur, chlorine and remaining heavy metals content of 0.47-0.64% was determined (Table 5.11; Table 5.12). A presumed average of 0.55% can be expected to be separated as air pollution control [APC] residues during the gas treatment. The remaining DFG was calculated as the syngas output (Table 7.7).

Table 7.7: In- and output flows for gasification

Name		Mixed DFG	Unit	Trawl nets	Gillnets	Background data
Input						
	DFG_3	909	kg	926	893	Results from Shredding
Output						
	Lead	67.6	kg	0	135	Lead separation efficiency = 100%
	Residues	419	kg	563	275	Mineral separation efficiency = 100%
	APC residues	5.00	kg	5.09	4.91	DFG_3 separation efficiency = 0.55%
	Syngas	417	kg	357	477	Calculated based on mass balance

Avoided production

The syngas output is calculated as 0.351-0.486 m^3/kg (Table 7.9). It assumes a syngas yield of 38.6-53.3% (based on Table 7.7) and a syngas density of 1.1 kg/m^3 (Hee, Horst and Quicker, 2018). It is modelled to substitute a biomass syngas production at a ratio of 1:1. While several feedstocks including coal and methane can serve as feedstock for the syngas production, biomass was selected due to its data availability in the Ecoinvent dataset.

Ancillary products and energy

The water consumption is calculated as 0.333 L/kg DFG assuming a water solid ratio of 25:75.

The nitrogen consumption is modelled as 40.8 g/kg dry DFG. This is because 16.9 kg of nitrogen were required for the steam gasification of 311 kg DFG with a water content of 25% (F. Rupert, personal communication, 29 January 2018).

The electricity consumption was determined as 1.65 kWh/kg dry DFG (Table 7.9). It is based on the electricity consumption of 384 kWh (Table 7.4) for the processing of 311 kg DFG with a water content of 25%.

The lime consumption is calculated as 3.25-4.98 g/kg DFG (Table 7.8). It is based on stoichiometric reactions between the lime and the contained chlorine and sulphur content of 0.3-0.43% and 0.05-0.09% respectively (Table 5.11).

Table 7.8: Calculation of lime consumption during gasification

Reaction	Molar ratios		Waste content [g/kg]		CaO need [g/kg]	
$H_2S + Ca(OH)_2 \rightarrow CaS + 2\ H_2O$	CaO / S	1.75	S	0.5-0.9	0.875-1.57	3.25-4.98
$2HCl + Ca(OH)_2 \rightarrow CaCl_2 + 2H_2O$	CaO / 2Cl	0.791	Cl	3.0-4.3	2.37-3.40	

An additional water consumption of 5-20 g/kg, a hydrochloric acid consumption of 0.625-2.5 mg/kg and a lime consumption of 3.13-12.5 mg/kg are needed to fill the two 4m^3 acid and basic scrubber tanks. This is part of 1-4 annual setup and maintenance processes and assumes a DFG throughput of 1600 t/a (F. Rupert, personal communication, 18 March 2019).

Table 7.9: Life cycle inventory for the steam gasification of 1kg input material

Name	Value	Unit	Probability distribution				Background data
			T	Min		Max	
			Log	SD	BU	Pedigree Score	
Avoided Production							
Syngas	0.418	m^3	T	0.351		0.486	38.6-53.5%; 1.1 kg/m^3
Ancillary products and energy							
Operation							
Water	0.333	kg	Log	1.3	1.05	4,5,1,1,1	Water to DFG ratio = 25:75
Nitrogen	40.8	g	Log	1.21	1.05	1,5,1,1,1	16.9 kg, 311 kg, 25% water
Electricity	1.65	kWh	Log	1.21	1.05	1,5,1,1,1	384 kWh, 311 kg, 25% water
Lime	4.11	g	T	3.25		4.98	Table 7.8
Setup and maintenance							
Water	12.5	g	T	5		20	8-32 m^3/a, 1600 t/a
HCL	1.56	mg	T	0.625		2.5	1-4 kg/a, 1600 t/a
Lime	7.81	mg	T	3.13		12.5	5-20 kg/a, 1600 t/a

7.2.4 Critical aspects

Syngas quantity

The syngas quantity does not consider the water input of 25% which will have formed the syngas output. Therefore, a sensitivity analysis is conducted in Chapter 10.3 to evaluate the effect of this uncertainty.

Syngas quality

The syngas had a very high hydrogen and carbon monoxide content so that a high heating value of up to 15 MJ/m^3 can be expected (Lopez et al., 2018). Also, the methane content was relatively low which indicates a reduced tar formation (Ibid). Another indication for a high syngas quality is the ratio of two between the hydrogen and carbon monoxide content. In fact, if the methane and carbon dioxide content can be reduced to below 3% in a next step, the syngas would be suitable for a Fischer-Tropsch synthesis as part of a chemical recycling (Basini, 2005). In comparison with an average syngas production from biomass, the syngas from the steam gasification is expected to have superior properties. A one to one substitution may thus not adequately capture the value so that a higher ratio is considered as part of the sensitivity analysis in Chapter 10.3.

Energy consumption

The electricity consumption was determined as 1.65 kWh/kg. This is very similar to comparable gasification plants for mixed plastic waste for example 1.58 kWh/kg in Al-Salem, Evangelisti and Lettieri (2014) or 1.39 kWh/kg in Hellweg (2000). Due to the small variability, a sensitivity analysis is not conducted for this input parameter.

8 Steel and Lead recycling

8.1 Steel recycling

8.1.1 Life cycle inventory

DFG materials

Within the modelled waste treatment scenarios, the sorting and shredding step separate steel from DFG for recycling (Figure 8.1). The steel mainly comprises of metal chains and anchors (Figure 8.1) which are typically made of either galvanised or stainless steel. For the modelling it is assumed that DFG contains 100% of galvanised steel. This is because the removed steel showed clear signs of corrosion, which would not be expected for stainless steel (Figure 8.1).

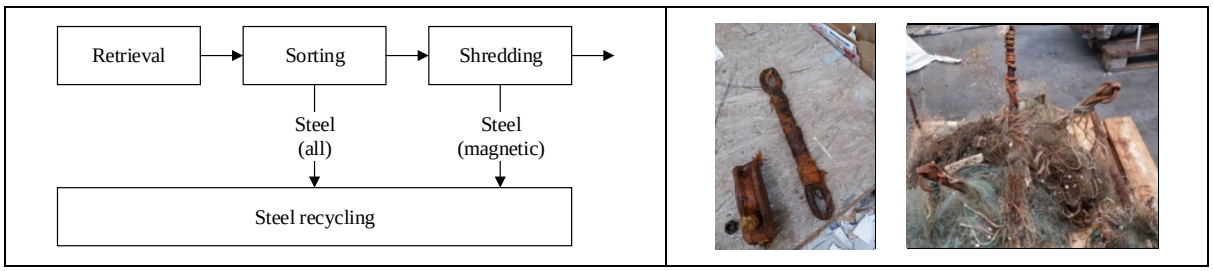


Figure 8.1: Separated steel from DFG

Technology selection

Large scale steel recycling either takes place in basic oxygen furnaces (BOFs) or in electric arc furnaces (EAFs). BOFs use pure oxygen to turn iron ore, coal and other additives into a comparatively pure and malleable steel. This makes it suitable for cold working applications such as for chassis in the automotive industry. They accept up to 30% of scrap (World steel Association, 2019), but the burning of coal and the preceded iron ore mining are linked to increased emissions. EAFs on the other hand, use electricity to melt up to 100% of scrap into new steel. It is possible that impurities such as copper accumulate which may inhibit its use for cold working applications.

The steel recycling is modelled via an electric arc furnace to reflect the most common pathway worldwide (World steel Association, 2019).

Process description

The process entails a pre-treatment, smelting and refinement as well as a finish, off-gas and residues treatment (Figure 8.2).

The pre-treatment is usually carried out by a scrap recycling facility and involves the collection, sorting, shredding and compaction of the scrap metal (Brooks et al., 2019). The smelting produces liquid steel by applying high temperatures of up to 3500°C. To influence the steel grade, alloys and slag formers are added into the smelter. The liquid steel is refined to adjust its exact chemical composition. Again, this can entail the addition of various alloys and compounds, for example to reduce the oxygen, carbon and sulphur content. To finish the process, a combination of casting, coating and rolling steps are applied.

During the smelting and refinement, off-gas and slag form. The slag is removed, solidified and landfilled or given to the construction industry (Remus et al., 2013). The off-gas is collected, freed from dust and released to the atmosphere. The dust is typically landfilled, possibly after a zinc recovery took place.

Included activities

The steel recycling process steps are illustrated in Figure 8.2. The pre-treatment, finish and landfilling of residues was excluded from the scope due to their expected lower environmental impact.

Included flows

The included flows are depicted in Figure 8.2. The consumption of process steam was excluded because it is only required for specific and less common vacuum treatments (Remus et al., 2013).

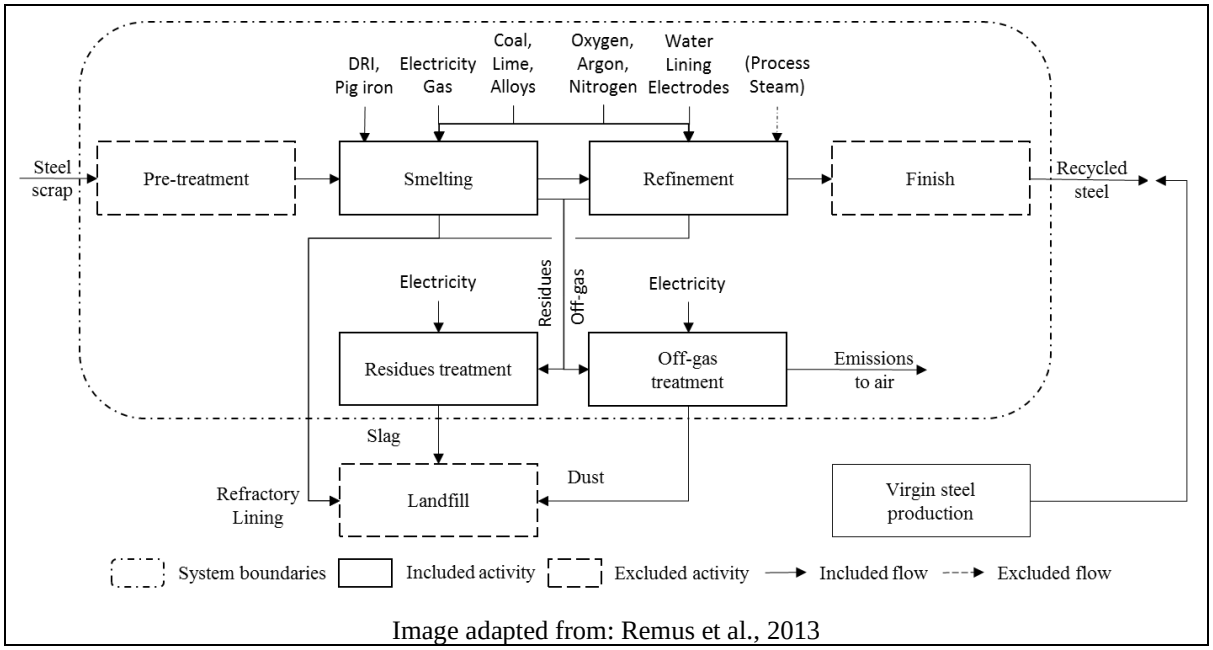


Image adapted from: Remus et al., 2013

Figure 8.2: LCI scope for the steel recycling

In- and output flows

The input steel scrap is based on the output steel from the sorting and shredding process (Table 5.8; Table 5.14).

The amount of recycled steel, smelting and refinement slag as well as dust and refractory lining was adopted based on average European data (Remus et al., 2013) and accounted for 81.2-96.2%, 4.9-26.0%, 0.8-7.7%, 0.8-2.9% and 0.1-2.2% of the steel scrap input respectively. For the calculations in Table 8.1, average conversion efficiencies were used.

The additional input flows (Figure 8.2) were summarised as ancillaries and determined based on the mass balance (Table 8.1).

Table 8.1: In- and output flows for steel recycling

Name	Mixed DFG	Unit	Trawl nets	Gill-nets	Background data
Input					
Steel scrap	73.7	kg	74.5	72.9	Calculated based on mass balance
Sorting	58.4	kg	59.0	57.7	Table 5.8
Shredding	15.4	kg	15.5	15.2	Table 5.14
Ancillaries	8.40	kg	8.49	8.31	Calculated based on mass balance
Output					
Recycled steel	65.4	kg	66.1	64.7	Steel conversion efficiency = 88.7% (81.2-96.2%)
Smelting slag	11.4	kg	11.5	11.2	Steel conversion efficiency = 15.4% (4.9-26.0%)
Refinement slag	3.17	kg	3.20	3.14	Steel conversion efficiency = 4.3% (0.8-7.7%)
Dust	1.33	kg	1.34	1.31	Steel conversion efficiency = 1.8% (0.8-2.9%)
Refractory lining	0.884	kg	0.894	0.875	Steel conversion efficiency = 1.2% (0.1-2.2%)

Avoided production

The amount of recycled steel was calculated as 0.812-0.962 kg/kg DFG (adopted from Remus et al., 2013). It was assumed that the recycled steel replaces 100% of average low-alloyed steel in the market.

Ancillary products and energy

The ancillary products and energy consumption are based on the best available techniques in Europe (Remus et al., 2013). The data was converted to represent the functional unit of 1 kg of steel scrap.

For a unit conversion a density of 1.429 kg/m^3, 1.784 kg/m^3 and 1.251 kg/m^3 was used for oxygen, argon and nitrogen respectively (Engineering ToolBox, 2003). For natural gas a heating value of 40 MJ/m^3 was assumed (Ibid).

Emissions to air

The airborne emissions are based on (Remus et al., 2013). As above, the data was adjusted to reflect the functional unit of 1 kg of steel scrap input.

Table 8.2: Life cycle inventory for the steel recycling of 1 kg input material

Name	Value	Unit	T	Min		Max	Background data	
			Log	SD	BU	Pedigree Score		
Avoided production								
Low-alloyed steel	0.887	kg	T	0.812		0.962	Scrap metal: 1039-1232 kg/t LS	Remus et al., 2013
Ancillary products and energy								
Pig iron	73.6	g	T	0		147	0 – 153 kg/t LS	Remus et al., 2013
Direct reduced iron	103	g	T	0		207	0 – 215 kg/t LS	
Lime	77.5	g	T	20.3		135	25-140 kg/t LS	
Charcoal	14.7	g	T	2.44		26.9	3-28 kg/t LS	
Graphite electrodes	3.70	g	T	1.62		5.77	2-6 kg/t LS	
Refractory lining	30.5	g	T	3.25		57.7	4-60 kg/t LS	
Alloys	23.7	g	T	8.93		38.5	11-40 kg/t LS	
Oxygen	47.6	g	T	5.80		89.4	5-65 m^3/t LS	
Argon	1.46	g	T	0.434		2.49	0.3-1.45 m^3/t LS	
Nitrogen	7.63	g	T	0.812		14.4	0.8-12 m^3/t LS	
Electricity	0.524	kWh	T	0.328		0.720	404-748 kWh/t LS	
Natural gas	18.6	dm^3	T	1.01		36.1	50-1500 MJ/t LS	
Water	21.0	kg	T	0.812		41.2	1-42.8 m^3/t LS	
Emissions to air								
Carbon dioxide	116	g	T	58.4		173	72-180 kg/t LS	Remus et al., 2013
Particulates	146	mg	T	3.25		289	4-300 g/t LS	
Nitrogen oxides	227	mg	T	10.6		443	13-460 g/t LS	
HCl	17.3	mg	T	0.649		33.9	800-35250 mg/t LS	
Sulphur dioxide	103	mg	T	4.06		202	5-210 g/t LS	
Carbon monoxide	2186	mg	T	40.6		4331	50-4500 g/t LS	
Zinc	11.6	mg	T	0.162		23.1	200-24000 mg/t LS	
Hydrogen fluoride	7218	ug	T	0.0325		14437	0.04-15000 mg/t LS	
Benzene	2130	ug	T	24.4		4235	30-4400 mg/t LS	
Chlorobenzenes	5.86	ug	T	0.162		11.5	0.2-12 mg/t LS	
PAH	470	ug	T	7.31		934	9-970 mg/t LS	
Mercury	97.1	ug	T	1.62		192	2-200 mg/t LS	
Lead	1402	ug	T	60.9		2743	75-2850 mg/t LS	
Chromium	1352	ug	T	9.74		2695	12-2800 mg/t LS	
Nickel	964	ug	T	2.44		1925	3-2000 mg/t LS	
Cadmium	71.6	ug	T	0.812		142	1-148 mg/t LS	
Copper	250	ug	T	8.93		491	11-510 mg/t LS	
PCB	2410	ng	T	8.12		4812	0.01-5 mg/t LS	
PCDD/F	2904	pg	T	32.5		5775	0.04-6 ug/t LS	

LS = Liquid steel, Alloys and low-alloyed steel = unalloyed steel, Direct reduced iron = pig iron

8.1.2 Critical aspects

DFG materials

It was assumed that DFG comprises of magnetic low-alloyed steel because the experiments revealed steel with clear signs of corrosion. Still, it is possible that high-alloyed steel is also

contained in DFG. In this case, large pieces would also be separated during the sorting step whereas smaller and potentially non-magnetic pieces may not be recycled at all. Although such considerations were not modelled here, future studies could investigate their possible impact on the results.

Steel output

For the steel recovery a conversion efficiency of 88.7% was assumed. This is quite a conservative estimate as the comparison with other studies shows. For example, Tunc, Camdali and Arasil (2012) determine the steel output as 95% and Pfeifer and Kirschen (2002) as 96.5%. Still, the selected data for the LCI based on Remus et al. (2013) represents more than 50% of the EAFs in Europe which makes it the most representative dataset.

Ancillary products and energy

The data used to model the ancillary products and energy consumption was compared with data from other studies (i.e. Lee and Sohn, 2014; Yetisken, Camdali and Ekmekci, 2013). As no significant anomaly was detected, no further uncertainty analysis was conducted.

8.2 Lead recycling

8.2.1 Life cycle inventory

DFG materials

Within the modelled waste treatment scenarios, lead is separated during the first density separation and gasification (Figure 8.3). It is recovered with little contamination in its metallic form (Figure 8.3). As such, it is comparable with other lead scrap such as sheets or piping from the construction sector.

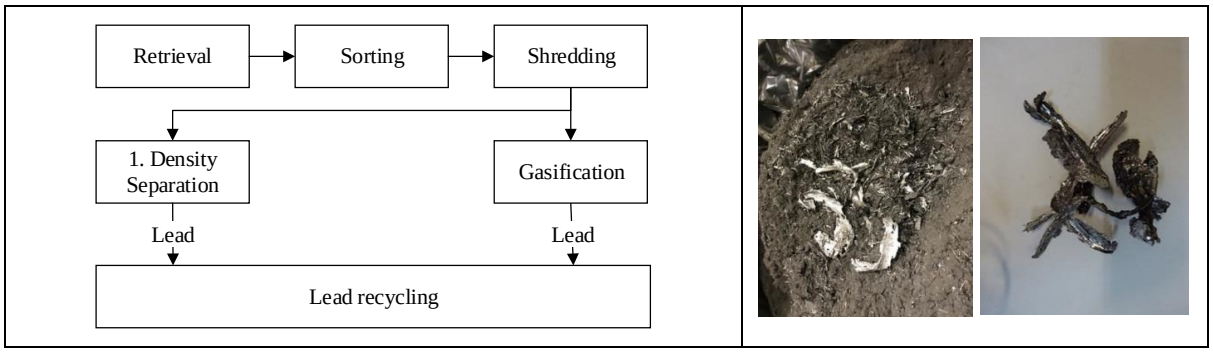


Figure 8.3: Separation pathway of lead (left) and experimental lead residues (right)

Technology selection

Lead recycling can be divided into pyro- or hydrometallurgical processes. Pyrometallurgical processes apply high temperatures to melt and refine lead. They are well established but also cause undesired airborne emissions (Thornton, Rautiu and Brush, 2001). Hydrometallurgical processes use chemicals to dissolve, purify and recover lead at lower temperatures. Although they are not yet established in Europe (Cusano et al., 2017), their lower airborne emissions may help hydrometallurgical processes to play an important role in the future (Thornton, Rautiu and Brush, 2001).

The pyrometallurgical lead recycling is either conducted in primary or secondary production facilities. Primary production facilities make lead from ore and only accept a small percentage of lead scrap while secondary production facilities accept 100% of lead scrap, particularly in form of lead-acid batteries (Thornton, Rautiu and Brush, 2001).

The lead recycling is modelled as part of a secondary production facility with a rotary furnace which represents a typical situation in Europe (Cusano et al., 2017).

Process description

The lead recycling includes a pre-treatment, smelting and refinement as well as a finish, gas and wastewater treatment.

During the pre-treatment lead scrap is collected, sorted and crushed before an acid neutralisation, desulphurisation and drying takes place (Thornton, Rautiu and Brush, 2001). Although, all steps are required for lead acid batteries, the pre-treatment would be skipped for the comparatively pure lead from DFG. Afterwards, the scrap lead is smelted into a lead bullion at temperatures of 1000°C-1200°C. During the refinement contaminants such as copper, silver and bismuth are removed. The purified lead is then mixed with alloys to adjust its desired chemical composition. Ultimately, an ingot casting finishes the process (Cusano et al., 2017).

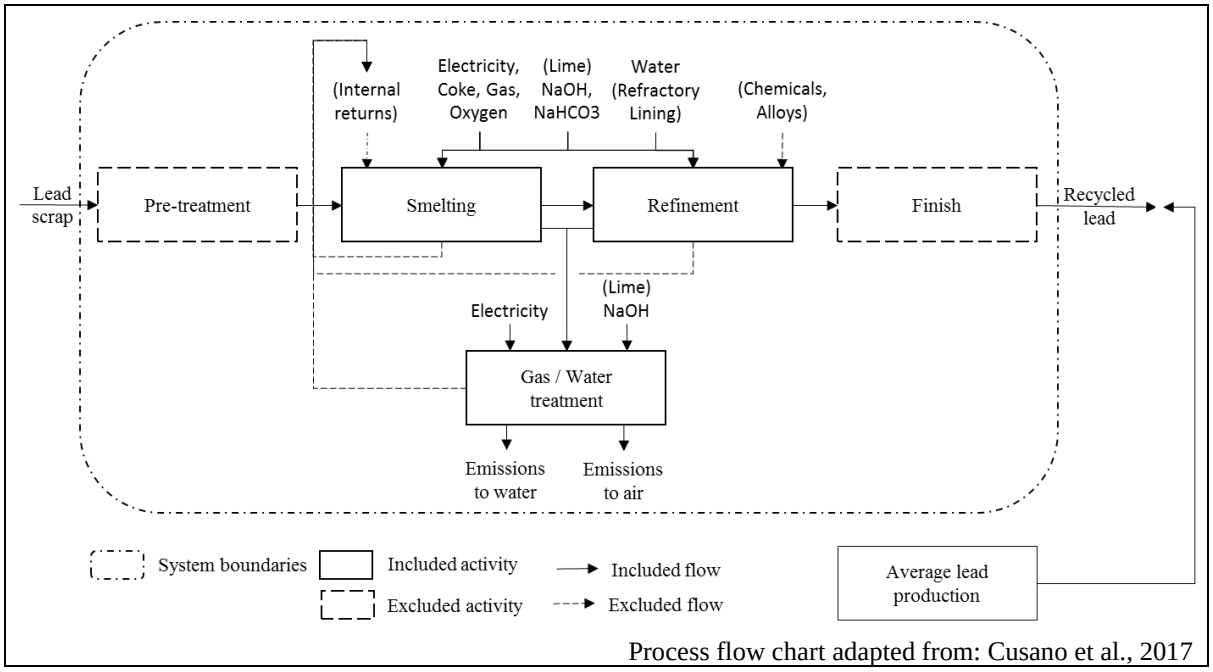
The smelting and refinement also generate flue gas, slag and wastewater. The flue gas is dedusted, stripped from remaining emissions and released to the atmosphere. The dust and slag are usually returned to the smelter (Cusano et al., 2017; Davidson, Binks and Gediga, 2016) whereas the wastewater is treated and released to the surface water.

Included activities

The lead recycling process steps are illustrated in Figure 8.4. The pre-treatment and finish were excluded due to their expected low environmental impact.

Included flows

The in- and output flows are depicted in Figure 8.4. As dust and slag are returned internally, they are not further considered. Other flows such as lime, refractory lining and alloys were also omitted because no appropriate data could be obtained.



Process flow chart adapted from: Cusano et al., 2017

Figure 8.4: LCI scope for the lead recycling

In- and output flows

The lead input is based on the lead output from the first density separation and gasification (Table 6.7; Table 7.7). It was assumed that 100% of the scrap input is turned into recycled lead output, because no significant material loss is expected to occur during the process (Davidson, Binks and Gediga, 2016; Table 8.3).

Table 8.3: In- and output flows for lead recycling

Name	Mixed DFG	Unit	Trawl nets	Gillnets	Background data
Input					
Lead scrap	67.6	kg	0	135	Table 6.7; Table 7.7
Output					
Recycled lead	67.6	kg	0	135	Lead conversion efficiency = 100%

Avoided production

The recycled lead is modelled to replace 100% of average lead material at the market. This is because it leaves the recycling process with virgin like properties.

Ancillary products and energy

The ancillary products and energy consumption are modelled based on average production data (Table 8.4) from a typical European lead recycling facility with an annual lead throughput of 43 thousand tonnes (Cusano et al., 2017). The modelled water input reflects the facility's wastewater output. For the unit conversion a density of 1.429 kg/m^3 was assumed for oxygen in Table 8.4.

Emissions to air

The carbon dioxide emissions were based on the modelled lead recycling facility (Cusano et al., 2017). Other airborne emissions (Table 8.4) were derived from the EMEP/EEA air pollutant emission inventory guidebook (2016c).

Emissions to water

The emissions to water in Table 8.4 were calculated by multiplying the waste water quantity of the modelled lead recycling facility (Cusano et al., 2017) with a typical waste water concentration from various European lead and tin production facilities.

8.2.2 Critical aspects

DFG materials

Unlike lead acid batteries, DFG lead is pure and already in its metallic form. Therefore, it requires fewer processing steps, ancillary products and energy while also causing less emissions to water and air. However, for the LCI model average data was used. This means that the potential environmental impact is overestimated and represents a worst-case.

LCI data

The modelled LCI was compared with a study from the German Environment Agency. While most data were in the same order of magnitude, the cadmium and particulate emissions to air were 84 and 11 times higher in the study (Giegrich, Liebich and Fehrenbach, 2007). This may be explained by less restrictive emission levels in the past. Still, as otherwise no major difference could be identified, the LCI model proved very robust.

Table 8.4: Life cycle inventory for the lead recycling of 1 kg input material

Name	Value	Unit	T	Min		Max	Background data	
			Log	SD	BU	Pedigree Score		
Avoided production								
Lead	1.00	kg	T	1.00		1.00	Table 8.3	
Ancillary products and energy								
Sodium hydroxide	163	g	Log	1.79	1.05	5,4,5,1,1	7000 t / 43000 t	Cusano et al., 2017
Sodium bicarbonate	51.2	g	Log	1.79	1.05	5,4,5,1,1	2200 t / 43000 t	
Charcoal	55.8	g	Log	1.79	1.05	5,4,5,1,1	2400 t / 43000 t	
Oxygen	0.123	kg	Log	1.79	1.05	5,4,5,1,1	3.7 MNm^3 / 43000 t	
Natural gas	32.6	dm^3	Log	1.79	1.05	5,4,5,1,1	1.4 MNm^3 / 43000 t	
Electricity	88.4	Wh	Log	1.79	1.05	5,4,5,1,1	3.8 GWh / 43000 t	
Water	2.44	kg	Log	1.79	1.05	5,4,5,1,1	105000 m^3 / 43000 t	
Refractory lining	0.814	g	Log	1.79	1.05	5,4,5,1,1	35 t / 43000 t	
Emissions to air								
Carbon dioxide	0.216	kg	Log	1.79	1.05	5,4,5,1,1	10800 t / 43000 t	
Particulates	8	mg	T	4.5		14	4.5-14 g/t	EEA 2016c
Sulphur oxides	5	g	T	4		6	4000-6000 g/t	
Lead	1.1	mg	T	0.5		2.5	0.5-2.5 g/t	
Cadmium	0.05	mg	T	0		0.1	0-0.1 g/t	
Arsenic	0.3	mg	T	0.15		0.5	0.15-0.5 g/t	
Zinc	0.05	mg	T	0		0.1	0-0.1 g/t	
PCB	2.6	ng	T	1.3		5.2	1.3-5.2 ug/t	
Dioxins / Furans	3.2	ng	T	1.1		9.6	1.1-9.6 ug/t	
Emissions to water								
Lead	0.193	mg	T	0		1.22	< 0.001-0.5 mg/L	Cusano et al. 2017
Zinc	0.415	mg	T	0		4.88	< 0.005-2 mg/L	
Cadmium	0.122	mg	T	0		1.71	< 0.001-0.7 mg/L	
Arsenic	92.3	ug	T	2.44		415	0.001- 0.17 mg/L	
Copper	27.6	ug	T	0		1221	0-0.5 mg/L	
Nickel	304	ug	T	17.1		6031	0.007-2.47 mg/L	
Antimony	0.342	mg	T	0		1.27	< 0.01-0.52 mg/L	
Suspended solids	24.4	mg	T	7.33		61.0	3-25 mg/L	
Silver	1.22	ug	T	0		2.44	≤ 0.001 mg/L	
Mercury	24.4	ug	T	0		58.6	n.d.-0.024 mg/L	
Iron	0.440	mg	T	0.125		2.93	0.051-1.2 mg/L	
Nitrogen	19.5	mg	T	0		19.5	n.d.-8 mg/L	
Phosphorus	0.122	mg	T	0		0.122	n.d.-0.05 mg/L	
AOX	1.59	mg	T	0		7.08	< 0.05-2.9 mg/L	
Hydrocarbons	2.44	mg	T	0		2.44	n.d.-1 mg/L	
COD	56.0	mg	T	6.10		107	2.5-44 mg/L	
Thallium	0.537	mg	T	0		2.03	< 0.01-0.83 mg/L	
Tin	342	ug	T	73.3		757	0.03-0.31 mg/L	

9 Incineration, Landfill and Transport

9.1 Incineration

9.1.1 Life cycle inventory

DFG materials

Within the modelled DFG waste treatment scenarios, three materials are incinerated. This includes the DFG_4 output from the first density separation, the floating fraction of the second density separation and the screen fraction from the washing process (Figure 9.1). All materials were previously sorted and shredded to avoid machine entanglements, fire hazards and inhomogeneous burning from otherwise long fibres and bulky waste (M. Teder, personal communication 13 November 2018).

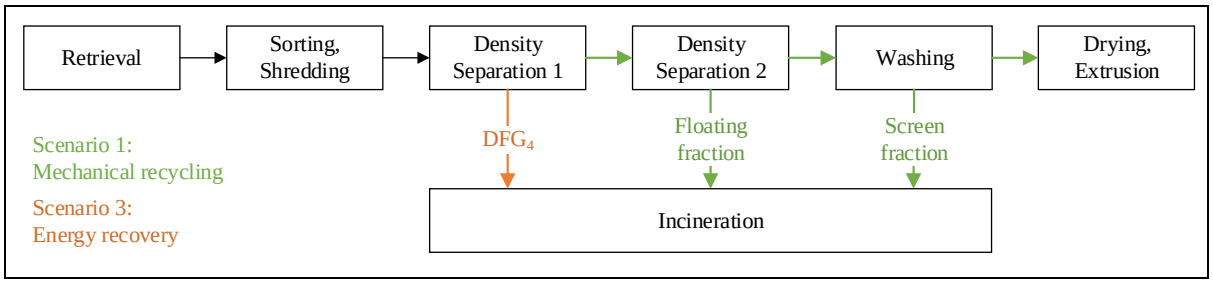


Figure 9.1: DFG materials for the incineration

Technology selection

For the LCI it is assumed that DFG materials are treated in an average European incineration plant for municipal solid waste [MSW].

MSW incinerators have different reactor designs, forms of energy recovery and flue gas treatments. To reflect a typical European technology setup (BREF, 2006) a moving grate incinerator with a joint heat and electricity production, a wet scrubbing and selective catalytic reduction was modelled.

Process description

The incineration process consists of a preparation, combustion and energy recovery as well as a flue gas, wastewater and solid residue treatment (Figure 9.2).

During the preparation, the DFG input is presumably washed, dried and homogenised to meet the required chlorine content and heating values. The DFG is then combusted into a hot flue gas and a bottom ash (BREF, 2006). The flue gas is used to raise steam for a district heating

and to produce electricity via a steam engine. It is then cleaned from the fly ash and passed through wet scrubbers for a heavy metal and acidic gas removal. Afterwards, the flue gas is freed from dioxins typically by mixing it with activated carbon and passing it through a fabric filter. The flue gas is then heated and mixed with ammonia to reduce its nitrogen oxides in a catalysator, before it is ultimately released via a stack.

The wastewater from the flue gas treatment is neutralised and filtered (BREF, 2006). To meet the legal requirements, it can be subject to further treatment steps before it is released to the surface water. The bottom ash is typically water cooled and separated from metals. The recovered metals are recycled whereas the remaining bottom fraction is either used in the construction sector or landfilled. The boiler ash and the air pollution control [APC] residues including the fly ash, loaded carbon and the wastewater treatment residues are generally given to a hazardous waste landfill.

Included activities

The incineration process is depicted in Figure 9.2. For the LCI the residues treatment was not considered because metals were not expected in the pre-treated DFG material. The landfilling of the residues is excluded due to their expected low impacts as discussed in Chapter 4.2.3.

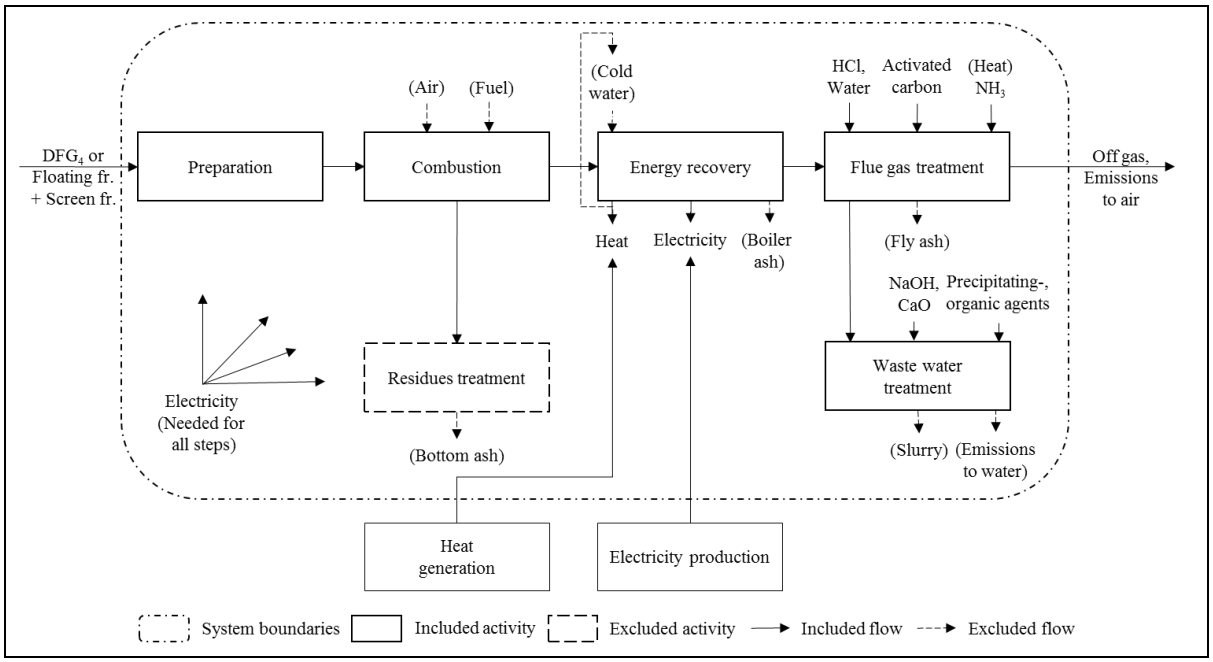


Figure 9.2: LCI scope for the incineration

Included flows

The in- and output flows are also illustrated in Figure 9.2. The air and cold water were not modelled due to their expected low environmental impact. The fuel consumption was excluded

because additional fuel is not required to burn the high calorific DFG material. For the selective catalytic reduction, the use of ammonia was included but the supplied heat was excluded.

In- and output data

For the energy recovery scenario, the DFG_4 input is based on the DFG_4 output from the first density separation (Table 6.7). The bottom ash equals the amount of the remaining mineral fraction in DFG_4 assuming a separation efficiency of 100%. The boiler ash and the air pollution control [APC] residues, were estimated as 7.0% based on the heavy metal, chlorine and sulphur content in dry DFG_4 (Table 6.3). The off gas resulted from the mass balance (Table 9.1).

Table 9.1: In- and output flows for incineration S_3

Name	Mixed DFG	Unit	Trawl nets	Gillnets	Background data
Input					
DFG_4	464	kg	419	510	Table 5.7
Output					
Bottom ash	41.9	kg	56.3	27.5	Mineral separation efficiency = 100%
APC residues + boiler ash	32.5	kg	29.3	35.7	DFG_4 separation efficiency = 7.0%
Off gas	390	kg	333	446	Calculated based on mass balance

For the mechanical recycling scenario, the floating and screen fraction derived from Table 6.8 and Table 6.14. It was assumed that no bottom ash is generated because no minerals remained in those two fractions. Based on the heavy metals, chlorine and sulphur content in the floating fraction (Table 6.3), the boiler ash and the APC residues were estimated at 0.8% of dry DFG input. The water and off gas were calculated based on the mass balance (Table 9.2).

Table 9.2: In- and output flows for incineration S_1

Name	Mixed DFG	Unit	Trawl nets	Gillnets	Background data
Input					
Floating fraction	23.2	kg	0.0	46.3	Table 6.8
Screen fraction	59.8	kg	54.3	65.4	Table 6.14
Output					
Bottom ash	0	kg	0	0	Mineral separation efficiency = 100%
APC residues + boiler ash	0.664	kg	0.435	0.894	Dry input separation efficiency = 0.8%
Off gas	82.4	kg	53.9	111	Calculated based on mass balance

Avoided production

The steam and electricity generation are calculated as 2.61-2.86 kWh/kg and as 1.06-1.16 kWh/kg respectively (Table 9.3). The calculation is based on the lower heating value of the DFG_4 and the floating fraction and assumes a typical European thermal and electrical efficiency of 37% and 15% (Laner et al., 2015). Other conversion efficiencies are further discussed below. The steam and electricity are assumed to replace an average electricity and heat production.

Table 9.3: Electricity and steam output calculation for incineration S_1 and S_3

Scenario	Material	LHV [MJ/kg]	Thermal Efficiency	Electrical Efficiency	Steam [kWh/kg]	Electricity [kWh/kg]
Energy Recovery (S3)	DFG_4	25.4 [a]	37% [b]	15% [b]	2.61	1.06
Mechanical Recycling (S1)	Floating fraction (Screen fraction)	27.8 [a]			2.86	1.16

[a] The values are based on Table 6.3. [b] The values are based on Laner et al. (2015).

Ancillary materials and energy

The sodium hydroxide and lime consumption were calculated as 26.7 g/kg and 38.1 g/kg for DFG_4 and as 2.72 g/kg and 3.89 g/kg for the floating fraction respectively (Table 9.4). This is based on the previously determined chlorine and sulphur content in the materials (Table 6.3) and the stoichiometric reactions for lime (Table 7.8) and sodium hydroxide (Table 9.5). It assumes that both neutralisation agents are used in 50% of the incinerators.

Table 9.4: Calculation of the sodium hydroxide and lime consumption

Scenario	Material	Waste content		CaO (50%)		NaOH (50%)	
		Element	[g/kg]	Molar ratio	[g/kg]	Molar ratio	[g/kg]
Energy Recovery (S3)	DFG_4	Cl	65.5	0.79	26.7	1.13	38.1
		S	0.9	1.75		2.50	
Mechanical Recycling (S3)	Floating fraction (Screen fraction)	Cl	4.9	0.79	2.72	1.13	3.89
		S	0.9	1.75		2.50	

Table 9.5: Stoichiometric reactions for sodium hydroxide

Reaction	Molar ratios	
$H2SO_4 + 2\ (NaOH) \rightarrow Na_2SO_4 + 2\ H_2O$	2 (NaOH) / S	2.50
$HCl + NaOH \rightarrow NaCl + H_2O$	NaOH / Cl	1.13

The remaining ancillary products and energy consumption were derived from the BREF (2006) (Table 9.7).

Emissions to air

The carbon dioxide emissions are calculated based on the carbon content and molar ratios as 1.06-2.03 kg/kg DFG (Table 9.6). It assumes a complete conversion of carbon into carbon dioxide. Furthermore, it was assumed that all carbon derived from fossil fuels (Table 9.6).

Table 9.6: Calculation of carbon dioxide emissions from incineration

Material	C [kg/kg]		C_{fossil} [%]	Molar ratio CO_2/C	CO_2 [kg/kg]		
	Min	Max			Min	Max	Average
DFG_4, Floating fraction, Screen fraction	0.290 [a]	0.554 [b]	100	3.66	1.06	2.03	1.55

[a] The value is based on the minimum carbon content measured in the shredding output (Table 5.11).
[b] The value is based on a typical carbon content in polyamide (Othman et al., 2008).

Table 9.7: Life cycle inventory for the incineration of 1 kg DFG material

Name	Value	Unit	T	Min		Max	Background data	
			Log	SD	BU	P. Score		
Avoided production								
Electricity	1.06 [a)]	kWh	Log	1.83	1.05	2,5,5,1,4	Table 9.3	
Heat from steam	2.61 [a)]	kWh	Log	1.83	1.05	2,5,5,1,4		
Ancillary materials and energy								
Lime	26.7 [a)]	g	Log	1.11	1.05	1,4,1,1,1	Table 9.4	
Sodium hydroxide	38.1 [a)]	g	Log	1.11	1.05	1,4,1,1,1		
Electricity	130	Wh	T	60		200	60-200 kWh/t	BREF,2006 (pp. 174, 203, 293, 368, 532, 555, 563)
Ammonia, 25%	2.75	g	T	0.5		5	0.5-5 kg/t	
Water	0.25	L	T	0.15		0.3	0.15-0.3 m^3/t	
Hydrochloric acid	188	mg	T	75		300	0.075-0.3 kg/t	
Activated carbon	0.75	g	T	0.5		1	0.5-1.0 kg/t	
Flocculation agents [b)]	2.23	g	T	0.003		4.45	0.003-4.45 kg/t	
Precipitating agents [c)]	0.15	g	T	0.05		0.25	0.05- 0.25 kg/t	
Emissions to air								
Carbon dioxide $_{fossil}$	1.55	kg	T	1.06		2.03	Table 9.6	
Nitrogen oxides	1071	mg	T	749		1532	749-1532 g/t	EEA, 2016d (Table 3-1)
Carbon monoxide	41	mg	T	7		253	7-253 g/t	
NMVOC	5.9	mg	T	2.7		12.9	2.7-12.9 g/t	
Sulphur dioxide	87	mg	T	16		466	16-466 g/t	
Ammonia	3	mg	T	0.5		18.3	0.5-18.3 g/t	
Particulates	3	mg	T	1.1		8.3	1.1-8.3 g/t	
Lead	58	ug	T	12		280	12-280.3 mg/t	
Cadmium	4.6	ug	T	1.1		19.3	1.1-19.3 mg/t	
Mercury	18.8	ug	T	7.3		48.3	7.3-48.3 mg/t	
Arsenic	6.2	ug	T	1.3		29.6	1.3-29.6 mg/t	
Chromium	16.4	ug	T	3		88.7	3-88.7 mg/t	
Copper	13.7	ug	T	3.9		47.3	3.9-47.3 mg/t	
Nickel	21.6	ug	T	4.2		112	4.2-111.6 mg/t	
Selenium	11.7	ug	T	2.2		62	2.2-62 mg/t	
Zinc	24.5	ug	T	2.7		220	2.7-219.6 mg/t	
PCBs	3.4	pg	T	1.2		9.2	1.2-9.2 ng/t	
Dioxins / furans	52.5	pg	T	16.6		166	16.6-166.3 ng/t	
Benzo[a]pyrene	8.4	ng	T	2.8		33.6	2.8-33.6 ug/t	
Benzo(b)fluoranthene	17.9	ng	T	6		71.4	6-71.4 ug/t	
Benzo(k)fluoranthene	9.5	ng	T	3.2		37.8	3.2-37.8 ug/t	
Indeno(1,2,3-cd) pyrene	11.6	ng	T	3.9		46.2	3.9-46.2 ug/t	
Hexachlorobenzene	45.2	ng	T	8		254	8-254.1 ug/t	
Hydrogen chloride	26.5 [d)]	mg	T	0.45 [d)]		60 [d)]	0.1-10 mg/m³	BREF, 2006 (p. 151)
Hydrogen fluoride	2.89 [d)]	mg	T	0.45 [d)]		6 [d)]	0.1-1 mg/m³	
Emissions to water								
Arsenic	11.6	ug	T	0		32.2	0-32.2 mg/t	BREF, 2006 (Table 3.26)
Cadmium	4.55	ug	T	0		9.1	0-9.1 mg/t	
Chromium	21.5	ug	T	0		43	0-43 mg/t	
Copper	57.5	ug	T	0		115	0-115 mg/t	
Mercury	1.52	ug	T	0		3.04	0-3.04 mg/t	
Lead	36	ug	T	0		72	0-72 mg/t	
Nickel	22.2	ug	T	0		44.4	0-44.4 mg/t	
Zinc	276	ug	T	0		552	0-552 mg/t	
Chlorides	2495	mg	T	0		4990	0-4990 g/t	
Sulphates	1035	mg	T	0		2070	0-2070 g/t	
COD	190	mg	T	0		380	0-380 g/t	
Nitrogen, total	65.5	mg	T	0		131	0-131 g/t	

[a)] The values represent scenario 3. [b)] The flow is modelled as aluminium hydroxide. [c)] The flow is modelled as cationic resign. [d)] The values assume an off gas volume of 4.5-6 m^3/kg waste (BREF, 2006). "P" - Pedigree

Other airborne emissions were determined with the Tier 1 emission factors for MSW incinerators (EEA, 2016d) or based on their average concentration in a presumed off gas volume of 4500-6000 m^3/t waste (BREF, 2006; Table 9.7).

Emissions to water

The emissions to water were based on data from Dutch incineration plants (BREF, 2006). The provided min- and max values were used to determine the average water emissions (Table 9.7).

9.1.2 Critical aspects

Solid residues

For the incineration it was assumed that all mineral content is ejected as bottom ash. However, some mineral content will enter the flue gas forming the boiler and fly ash (BREF, 2006). In fact, the ratio between the bottom ash and the boiler and fly ash is approximately 10:1 (Ibid). This is important because it determines the amount of hazardous and non-hazardous waste. However, in the current LCI model the ash treatment was not considered, so that a more accurate allocation of the mineral content has no effect. Still, this may be considered in future studies. Also, the contribution of ancillary materials such as activated carbon and neutralisation agents to the solid residues should then be included.

Energy output

The energy output is highly dependent on the thermal and electrical conversion efficiency of the incineration plant. In the baseline a thermal and electrical efficiency of 37% and 15% was assumed. However, the electrical efficiency can vary between 5-35% (Laner et al., 2015; BREF, 2006) and the thermal efficiency can lie between 3% and 72% (Laner et al., 2015). To account for this a sensitivity analysis is conducted in Chapter 10.3 with both min-max combinations.

Ancillary products

The modelled ancillary products were compared with other values from the literature (i.e. Beylot and Villeneuve, 2013). This revealed that the modelled water consumption may have been underestimated by a factor of three. As water was not considered a key contributor to environmental impacts, this difference was not further investigated. Other ancillary products had comparable values.

Emissions

The emissions to air and water are modelled based on average data for municipal solid waste. However, DFG may lead to higher emission values for example due to its elevated lead content. For a more conservative estimate, the maximum permittable emission limit values (Directive 2010/75/EU, 2010) may be used. However, like average data, the legal emission limits do not reflect the physical properties of DFG. For example, if chlorine was not contained in DFG, chlorine emissions should also not be included. To resolve this, transfer coefficients are sometimes used (i.e. Hellweg, 2000). They assign elements of a specific waste composition to different output flows. Thus, they provide a direct link with the input material and avoid the over- or underestimation of element flows. Transfer coefficients for incineration plants are provided in Laner et al. (2015). However, those were not applied in this study, because unlike average data, the heavy metals other than arsenic and mercury were not modelled to be emitted to air. Still, more precise transfer coefficients may be used in future studies.

9.2 Landfill

9.2.1 Life cycle inventory

DFG materials

In each waste treatment scenario for DFG, several fractions enter landfills (Figure 9.3). In this thesis only the landfilling of DFG_1 is discussed. The landfilling of other residues was excluded due to their expected low environmental impact (Chapter 4.2.3). DFG_1 contains 7.3-7.4% steel, 0-13.5% lead, 27.5-56.3% minerals and 36.2-48.2% polymers.

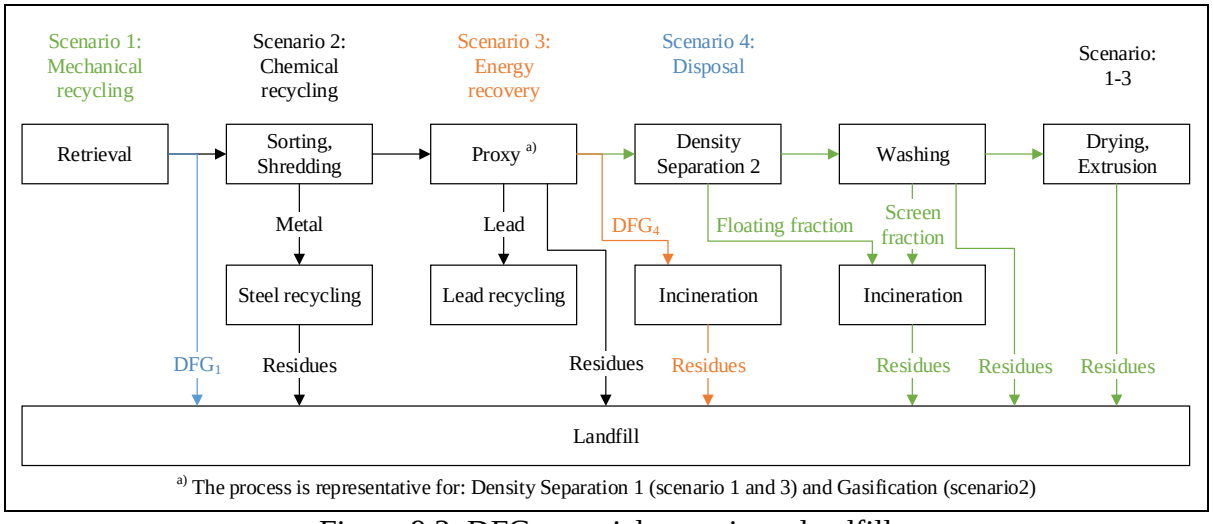


Figure 9.3: DFG materials entering a landfill

Technology selection

To reflect a typical landfill design in Europe, a closed landfill with a landfill gas [LFG] collection and utilisation as well as a leachate collection and treatment is modelled. For the LFG utilisation a combined heat and electricity recovery is assumed.

Process description

The landfill activities entail a pre-treatment, closure and landfill phase as well as the leachate and landfill gas treatment (Figure 9.4).

The pre-treatment for DFG can consist of a sorting, shredding and composting step to separate metals, lead and polymers as well as to reduce the total organic carbon in the waste. It is then placed in a landfill cell, compacted and covered to ensure its stability and a reduced rainwater penetration. The landfill gas and leachate are collected and treated over a specific period. The generated heat and electricity are typically used internally, and the cleaned leachate is released to the surface water.

Included activities

The landfill process steps are illustrated in Figure 9.4. To present a worst-case without a material recovery, the pre-treatment was not included in the LCI model.

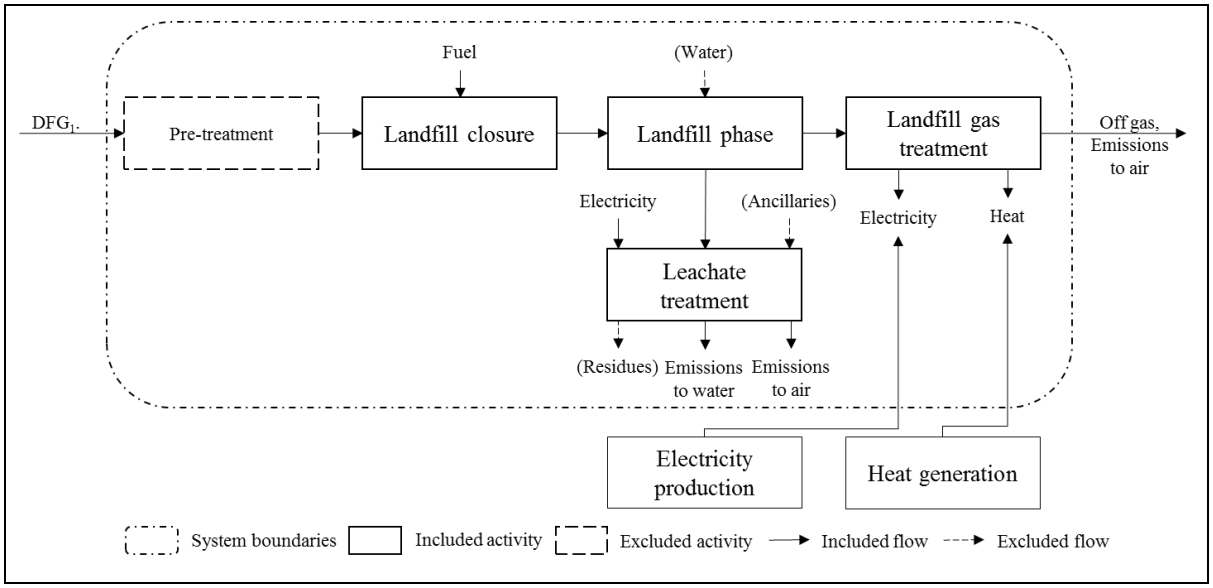


Figure 9.4: LCI scope for the landfill

Included flows

The considered in- and output flows are the (1) DFG_1 input and the (2) fuel, electricity and heat consumption and/or generation as well as the (3) emissions to water and air.

The rainwater, ancillary materials and residues were not included in the LCI. This is either due to a lack of data or their expected low environmental impact.

In- and output flows

The DFG_1 input is based on the DFG_1 output from the retrieval process (Table 5.4). There are no significant solid output flows from the modelled landfill.

Avoided production

The electricity and heat generation are calculated as 0.605-0.834 Wh/kg DFG and as 0.297-0.410 Wh/kg DFG respectively (Table 9.8). This is based on a 100-year period and assumes a typical carbon degradation rate of 1% in polymers (Rossi et al., 2015). The carbon content in DFG_1 was calculated by multiplying its polymer content with a carbon content of 55.4% from PA6 (Othman et al., 2008). Following the approach by Doka (2003a), 97.1% of the degraded carbon was estimated to transform into LFG, of which 49% was captured (Rossi et al., 2015) having a typical methane content of 56% (Doka, 2003a). For mass and energy conversions the molar ratios and a lower heating value of 50 MJ/kg for methane were applied. As thermal and electrical efficiencies, 3.0% and 6.1% were assumed respectively (Rossi et al., 2015).

Table 9.8: Calculation of the energy recovery from landfill gas

Poly-mer [%]	Car-bon [%]	Degra-dation [%]	LFG Fate [%]	LFG Capture [%]	LFG CH_4 [%]	Molar ratio [-]	LHV of CH_4 [MJ/kg]	Energy Type [-]	Effi-ciency [%]	Energy Generation [Wh/kg]
36.2-48.2	55.4	1.0	97.1	49.0	56	1.34	50.0	Thermal	3.0	0.297-0.396
								Electrical	6.1	0.605-0.805

Ancillary materials and energy

The fuel consumption in form of diesel was established as 0.879 g/kg DFG. This is based on an estimated consumption of 40 MJ/t waste (Koroneos and Nanaki, 2012) and an assumed heating value of 45.5 MJ/kg for diesel (European Automobile Manufacturer Association, n.d.).

The electricity consumption was estimated as 0.694 Wh/kg (Table 9.10). This is based on a leachate generation of 2.5 L/kg waste (Doka, 2003a) and an electricity demand of 1 kJ/L for the leachate treatment (Koroneos and Nanaki, 2012).

Emissions to air and water

As main elements, the carbon and lead related emissions are calculated in Table 9.9. For carbon, the same degradation, fate and capture rate as well as the LFG composition were assumed as in Table 9.8. The carbon content in the captured LFG was presumed to fully transform into

carbon dioxide during the energy recovery. The not captured LFG is directly emitted to air. Of the dissolved carbon, 85% are assumed to be captured and passed on to the leachate treatment (Koroneos and Nanaki, 2012). The leachate treatment removes 24.5% of the carbon as carbon dioxide and 65.8% as wastewater treatment residues (Doka, 2003b). The remaining 9.7% are released to the surface water.

Lead was assumed to lose 0.1% of its mass in form of leachate (Koroneos and Nanaki, 2012) and 0.033% as landfill gas (Doka, 2003a) within the first 100 years. A lead removal from combusted LFG is not expected so that all lead in LFG is emitted to the atmosphere. Like for carbon, a leachate collection of 85% is assumed (Koroneos and Nanaki, 2012). The leachate treatment is modelled to remove 90% of the lead as residues whereas the remaining 10% are emitted to the surface water (Doka, 2003b).

The landfilling of carbon or lead related sludge is not included in the LCI model.

Other airborne emissions were derived from average data for MSW (DEFRA, 2004) and adjusted to reflect an LFG capture and treatment of 49%.

From the same study, average leaching values were taken to calculate the emissions to water. A leachate volume of 2.5 L/kg (Doka, 2003a) and its complete capture and treatment were assumed for a period of 100 years. To avoid double counting, leaching values for carbon-based compounds were not considered.

Table 9.9: Calculation of carbon and lead related landfill emissions for 1kg of DFG_1

Element [%]		Degradation [%]		Fate [%]		Capture [%]		Treatment [%]		Mol. ratio [-]	Emission values Min-Max [mg/kg]	
C	20.1-26.7 [a)]	Year 0-100	1 [b)]	LFG	97.1 [b)]	N	51	CH_4	56.0 [b)]	1.34	743	989
								CO_2	44.0 [b)]	3.66	5144	6848
						Y	49 [c)]	CO_2	100			
				Leachate	2.9	Y	85 [d)]	CO_2	24.5 [e)]			
								LF	65.8 [e)]	1	-	-
								TOC	9.7 [e)]		13.5	18.0
						N	15	TOC	100			
Pb	0.0-13.5	Year 0-100	0.133 [f)]	LFG	24.8 [f)]	Y/N	100	Pb	100		0	44.6
				Leachate	75.2 [f)]	Y	15	Pb	100		0	32.9
							85 [d)]	Pb	10 [g)]			
								LF	90 [g)]		-	-

[a)] The values assume a carbon content of 55.4% in polymers based on Othman et al. (2008). [b)] The values are taken from the Ecoinvent landfill model (Doka, 2003a). [c)] The value is based on average European data from 2008 (Rossi et al., 2015). [d)] The value is based on European data (Koroneos and Nanaki, 2012). [e)] The value is based on Swiss data (Doka, 2003b). [f)] The value assumes a lead release of 0.033% as landfill gas (Doka, 2003a) and of 0.1% as leachate (Koroneos and Nanaki, 2012). [g)] The value assumes a lead removal of 90% during the leachate treatment (Doka, 2003b).

Table 9.10: Life cycle inventory for the landfilling of 1 kg input material

Name	Value	Unit	T	Min		Max	Background data	
			Log	SD	BU	Pedigree Score		
Avoided production								
Electricity	0.705	Wh	T	0.605		0.805	Table 9.8	
Heat	0.347	Wh	T	0.297		0.396		
Ancillary products								
Diesel	0.879	g	Log	1.64	1.05	4,5,3,1,4	40 MJ/t waste	Koroneos and Nanaki, 2012
Electricity	0.694 [a]	Wh	Log	1.64	1.05	4,5,3,1,4	1 kJ/L leachate	
Emissions to air								
Carbon dioxide	6.00	g	T	5.14		6.85	Table 9.9	
Methane	866	mg	T	743		989		
Lead	22.3	mg	T	0		44.6		
Nitrogen oxides	441	mg	T	196 [b]		1029 [b]	No data [c]	0.4-2.1 kg/t [d]
Particulates	3.43	mg	T	0.980 [b]		11.3 [b]	No data [c]	2-23 g/t [d]
Sulphur oxides	34.3	mg	T	14.7 [b]		83.3 [b]	No data [c]	30-170 g/t [d]
Hydrogen chloride	2.06	mg	T	0.456 [b]		11.3 [b]	0.03-1.1 g/t [c]	0.9-22 g/t [d]
Hydrogen fluoride	1.98	mg	T	0.444 [b]		9.95 [b]	5-300 mg/t [c]	0.9-20 g/t [d]
NMVOC	27.5	mg	T	9.06 [b]		106 [b]	12-55 g/t [c]	6-160 g/t [d]
1,1-dichloroethane	1377	ug	T	66.3 [b]		27030 [b]	0.13-53 g/t [c]	No data [d]
Chloroethane	510	ug	T	25.5 [b]		10710 [b]	0.05-21 g/t [c]	No data [d]
Chloroethene	561	ug	T	30.6 [b]		11220 [b]	0.06-22 g/t [c]	No data [d]
Chlorobenzene	1224	ug	T	61.2 [b]		24480 [b]	0.12-48 g/t [c]	No data [d]
Tetrachloroethene	1.78	mg	T	0.131 [b]		34.4 [b]	0.17-67 g/t [c]	0.9-0.4 g/t [d]
Cadmium	49.0	ug	T	9.80 [b]		294 [b]	No data [c]	0.02-0.6 g/t [d]
Nickel	6.37	ug	T	3.43 [b]		9.80 [b]	No data [c]	7-20 mg/t [d]
Arsenic	0.784	ug	T	0.392 [b]		1.47 [b]	No data [c]	0.8-3 mg/t [d]
Mercury	0.784	ug	T	0.392 [b]		1.47 [b]	No data [c]	0.8-3 mg/t [d]
Dioxins / furans	93.1	pg	T	6.86 [b]		1225 [b]	No data [c]	14-2500 ng/t [d]
Benzene	122	ug	T	6.12 [b]		2448 [b]	12-4800 mg/t [c]	No data [d]
Emissions to water								
TOC	15.8	mg	T	13.5		18.0	Table 9.9	
Lead	15.9	mg	T	0		32.9		
Chloride	2863	mg	T	1590 [a]		5153 [a]	1145 mg/L [e]	Factor = 1.8 [e]
Nitrogen	910	mg	T	455 [a]		1820 [a]	364 mg/L [e]	Factor = 2.0 [e]
Fluoride	1.63	mg	T	0.855 [a]		3.09 [a]	0.65 mg/L [e]	Factor = 1.9 [e]
Phosphorus	7.35	mg	T	4.08 [a]		13.2 [a]	2.94 mg/L [e]	Factor = 1.8 [e]
Organo-tin	0.500	ug	T	0.250 [a]		1.00 [a]	0.2 µg/L [e]	Factor = 2.0 [e]
Arsenic	6.00	ug	T	3.00 [a]		12.0 [a]	2.4 µg/L [e]	Factor = 2.0 [e]
Chromium	87.5	ug	T	43.8 [a]		175 [a]	35 µg/L [e]	Factor = 2.0 [e]
Copper	13.8	ug	T	4.58 [a]		41.3 [a]	5.5 µg/L [e]	Factor = 3.0 [e]
Nickel	120	ug	T	40.0 [a]		360 [a]	48 µg/L [e]	Factor = 3.0 [e]
Zinc	101	ug	T	11.3 [a]		911 [a]	40.5 µg/L [e]	Factor = 9.0 [e]

[a] The value assumes a leachate volume of 2.5 L/kg (Doka, 2003a). [b] The value is calculated as sum of 51% landfill gas emissions from a direct release (first column background data) and 49% landfill gas emissions from a gas engine (second column background data). [c] The value represents direct landfill gas emissions (DEFRA, 2004; Table 2.34). When "No data" was available, no emissions were assumed. [d] The value represents emissions from a gas engine (DEFRA, 2004; Table 2.36). When "No data" was available, no emissions were assumed. [e] The value or uncertainty factor represent emissions discharged to the surface water (DEFRA, 2004; Table 2.40).

9.2.2 Critical aspects

Technology selection

The landfilling of recyclable materials such as polymers and lead is increasingly restricted in Europe. For example, even hazardous waste landfills in Germany are not allowed to accept waste with more than 10% of lead (Versatzverordnung, 2002). Although there may be a gap between the legal requirements and practice, future studies may investigate the effect of introducing a pre-treatment step. To show the worst-case, a pre-treatment was intentionally excluded here.

Avoided production

The calculated energy generation is comparatively small. In fact, it is just enough to cover the internal electricity and possibly heat demand for the landfill activities (Table 9.10). This is mainly due to the slow degradation rate of polymers. Therefore, a major environmental benefit from the energy production cannot be expected.

Emissions to air and water

A major limitation of the LCI is the exclusion of long-term emissions that occur after 100 years. For example, 99.0% of the carbon and 99.9% of the lead related emissions are currently neglected (Table 9.9). Similarly, the arsenic or nickel emissions (Table 9.10) only represent 0.2% or 0.6% of their corresponding DFG content (Table 5.12). This means that the actual emission potential of landfills may be much greater than currently modelled. Still, long-term emissions are commonly excluded in other LCA studies due to their high uncertainty and the resulting dominance of heavy metals in toxic impact categories (Laurent et al., 2014a). Following this reasoning, a focus on the more certain short-term emissions was placed in this thesis as well.

Apart from carbon dioxide, methane and lead, the emissions to air and water are based on average data from MSW. Although the selected data (DEFRA, 2004) is in line with other studies (i.e. Di Maria, Sordi and Micale, 2013), it may not be directly applicable to DFG. This is mainly because the carbon related emissions such as chloroethane, tetrachloroethene or 1,1-dichloroethane result from an initially higher carbon availability in MSW compared to the slowly degrading DFG. This also explains the lower leaching values of fishing gear (Spadea et al., 2009) when compared with MSW (Table 9.10). In future studies, more appropriate datasets (i.e. Ibid) or a conversion factor may be applied to reflect the slower degradation rate.

9.3 Transport

9.3.1 Life cycle inventory

Transport routes

DFG materials require transportation between all treatment steps (Figure 9.5). This includes an external transportation via roads as well as an internal transportation through conveyor belts or forklifts. Due to the expected low environmental impact of the internal transport, only external transports were considered here (red arrows, Figure 9.5).

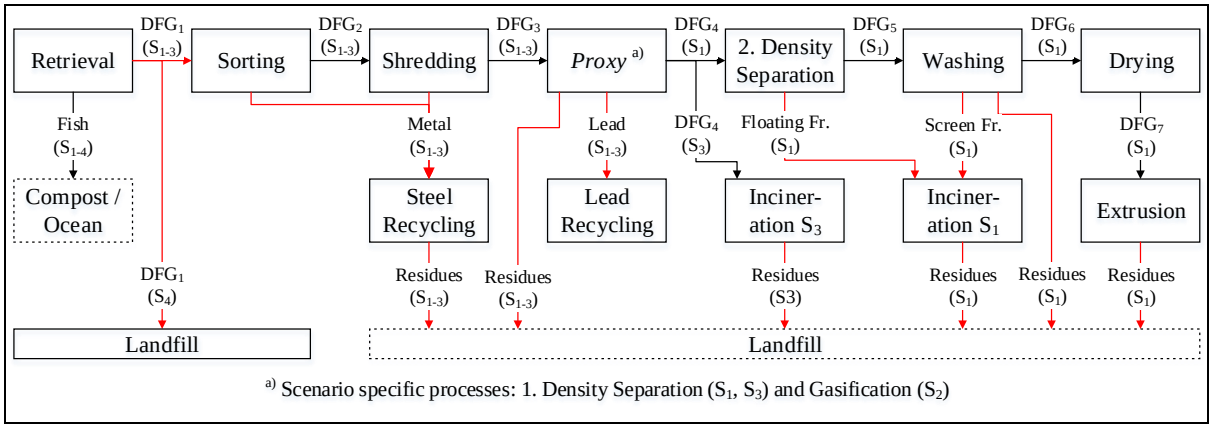


Figure 9.5: Overview of material transports (arrows) in the waste treatment scenarios

To model the transport, a centralised logistic approach with a main treatment plant for all primary processes was assumed. In this constellation, transports are needed between the harbour and the central treatment plant as well as to carry separated materials or residues to secondary plants or landfills (Figure 9.5).

Technology selection

For the transport a EURO 6 compliant heavy-duty diesel vehicle with a capacity of 3.5t – 7.5t was assumed. This is based on the experiments for which similar vehicles were used.

Included activities and flows

Apart from the actual transport, the process also involves the loading and unloading (Figure 9.6). While this may require energy or big bags, it can be expected that the impact from those tasks is relatively low. Thus, only the main transport process is modelled.

The transport's inventory data includes the lubricant oil and diesel consumption as well as airborne emissions from the combustion, brake and tyre wear (Figure 9.6).

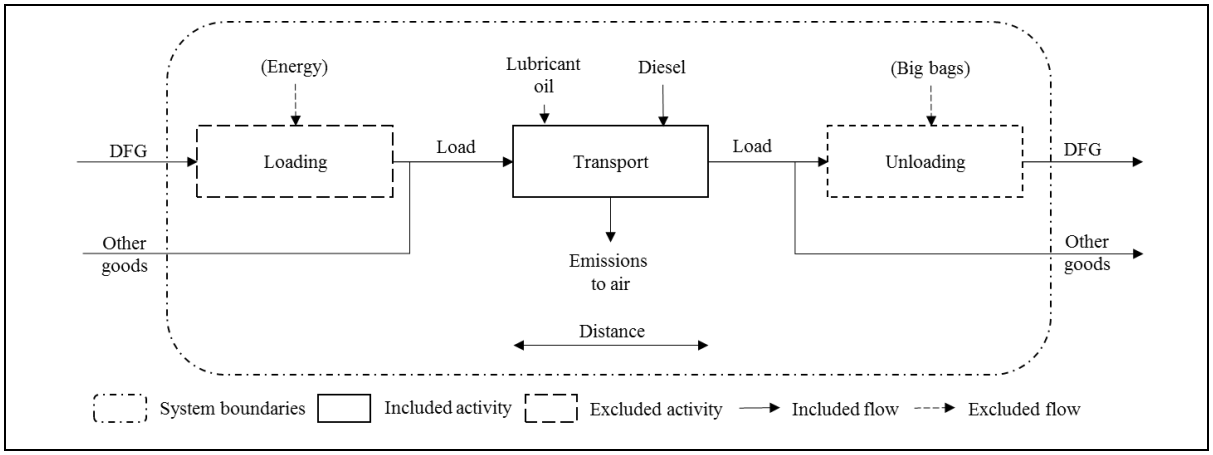


Figure 9.6: LCI scope for the transport

Transport model

Unlike previous processes, the inventory data of the transport process is modelled to be directly dependent on the transport distance. For example, to drive 100 km approximately 10.1 kg diesel would be consumed. However, as the truck load also influences the environmental impact the concept of a weighted transport distance is introduced. For this, the transport distance is multiplied with the material weight's contribution to a full load.

Weighted transport distance

Table 9.11 provides an overview of the weighted transport distances for each scenario. However, for the calculation (Table 9.12) a few assumptions needed to be made. Based on actual locations like Plastix Global in Denmark, a distance of 500 km to the central treatment plant; 250 km to the steel and lead recycling; as well as 20 km to the incineration and landfill were presumed. The assumed vehicle load of 0.98 tonnes was taken from the Ecoinvent database.

Table 9.11: Weighted transport distance for all scenarios

Name	Mixed DFG	Unit	Trawl nets	Gillnets	Background data
Scenario 1	548	km	542	554	See Table 9.12
Scenario 2	546	km	541	552	
Scenario 3	547	km	542	552	
Scenario 4	20.1	km	20.4	19.7	

Ancillary materials

The ancillary material consumptions were taken from the air pollutant and emission inventory guidebook (EEA, 2016b) as 101 g/km for diesel and as 1.56 kg/ 10,000 km for lubricant oil.

Table 9.12: Calculation of the weighted transport distance

Material	Distance [km]	Quantity [kg]			Weighted transport distance [km]							
					Trawl nets				Gillnets			
		Trawl nets	Gillnets	Load	S_1	S_2	S_3	S_4	S_1	S_2	S_3	S_4
DFG_1	500	1000	966	980	510	510	510		493	493	493	
	20							20.4				19.7
Metal	250	74.5	72.9		19.0	19.0	19.0		18.6	18.6	18.6	
Lead		0	135		0	0	0		34.5	34.5	34.5	
Floating + Screen	20	54.3	112		1.11				2.28			
Residues												
Steel recycling	20	16.9	16.6	980	0.35	0.35	0.35		0.34	0.34	0.34	
Den. Separation		507	248		10.3		10.3		5.06		5.06	
Gasification		568	280			11.6				5.72		
Incineration S_1		85.6	63.2				1.75				1.29	
Incineration S_3		0.435	0.894		0.01				0.02			
Washing		56.3	27.5		1.15				0.56			
Extrusion		5.54	6.67		0.11				0.14			
Total					542	541	542	20.4	554	552	552	19.7

Emissions to air

The emissions to air were divided into distance dependent emission, diesel consumption dependent emissions, lubricant oil consumption dependent emissions and emissions from the tyre and brake wear (Table 9.13). Like the ancillary materials, the emission data was taken from the EMEP/EEA guidebook (EEA, 2016b). The methane emission was selected to represent a highway situation, whereas the lead and sulphur dioxide emissions were calculated based on their relative mass in lubricant oil or diesel fuel. For the emission of lead a factor of 75% and for the transformation of sulphur into sulphur dioxide a factor of 200% was assumed (Ibid).

The tyre and brake wear (Table 9.12) were derived from the EMEP/EEA guidebook as well (EEA, 2016e).

9.3.2 Critical aspects

Logistic approach

It is also possible to establish a decentralised logistic network, where multiple pre-treatment steps are carried out at the harbour. To investigate the expected lower impact of such a system, a sensitivity analysis is conducted in Chapter 10.2.

Table 9.13: Life cycle inventory for the transport of DFG per km

Name	Value	Unit	Probability Distribution				Background data	
			T	Min		Max		
			Log	SD	BU	Pedigree Score		
Ancillary products and energy								
Diesel	101	g	Log	1.83	1.05	5,5,5,1,2	101 g/km	3-27
Lubricant oil	156	mg	Log	1.83	1.05	5,5,5,1,2	1.56E-04 kg/km	3-30
Emissions to air								
Distance dependent emissions								
Methane	20	mg	Log	2.07	1.5	5,5,5,1,2	20 mg/km	3-47
NMVOC	5.00	mg	Log	2.07	1.5	5,5,5,1,2	0.005 g/km	3.21
Carbon monoxide	47.0	mg	Log	5.58	5.0	5,5,5,1,2	0.047 g/km	
Nitrogen oxides	180	mg	Log	2.07	1.5	5,5,5,1,2	0.18 g/km	
Nitrous oxide	17.0	mg	Log	2.07	1.5	5,5,5,1,2	0.017 g/km	
Ammonia	9.00	mg	Log	2.07	1.5	5,5,5,1,2	0.009 g/km	
Carbon dioxide (lube)	486	mg	Log	1.83	1.05	5,5,5,1,2	0.486 g/km	
Particulates	500	ug	Log	2.51	2.0	5,5,5,1,2	0.0005 g/km	3-22
Indeno(1,2,3-CD)pyrene	1.40	ug	Log	2.07	1.5	5,5,5,1,2	1.40E-06 g/km	
Benzo(k)fluoranthene	6.09	ug	Log	2.07	1.5	5,5,5,1,2	6.09E-06 g/km	
Benzo(b)fluoranthene	5.45	ug	Log	2.07	1.5	5,5,5,1,2	5.45E-06 g/km	
Benzo(a)pyrene	900	ng	Log	2.07	1.5	5,5,5,1,2	9.00E-07 g/km	
Fuel consumption dependent emissions								
Carbon dioxide	320	kg	Log	1.83	1.05	5,5,5,1,2	3.169 kg /kg	3-29
Sulphur dioxide	606 [b)]	mg	Log	1.83	1.05	5,5,5,1,2	3 mg/kg	3-14
Lead	37.9 [c)]	ng	Log	5.58	5.0	5,5,5,1,2	0.0005 mg/kg	3-78
Zinc	1.82	ug	Log	5.58	5.0	5,5,5,1,2	0.018 mg/kg	
Cadmium	5.05	ng	Log	5.58	5.0	5,5,5,1,2	5E-05 mg/kg	
Copper	576	ng	Log	5.58	5.0	5,5,5,1,2	0.0057 mg/kg	
Chromium	859	ng	Log	5.58	5.0	5,5,5,1,2	0.0085 mg/kg	
Nickel	20.2	ng	Log	5.58	5.0	5,5,5,1,2	0.0002 mg/kg	
Selenium	10.1	ng	Log	5.58	5.0	5,5,5,1,2	0.0001 mg/kg	
Mercury	535	ng	Log	5.58	5.0	5,5,5,1,2	0.0053 mg/kg	
Arsenic	10.1	ng	Log	5.58	5.0	5,5,5,1,2	0.0001 mg/kg	
Lubricant oil dependent emissions								
Copper	121	ug	Log	5.58	5.0	5,5,5,1,2	778 mg/kg	3-79
Zinc	70.2	ug	Log	5.58	5.0	5,5,5,1,2	450.2 mg/kg	
Nickel	4.97	ug	Log	5.58	5.0	5,5,5,1,2	31.89 mg/kg	
Chromium	3.00	ug	Log	5.58	5.0	5,5,5,1,2	19.2 mg/kg	
Cadmium	711	ng	Log	5.58	5.0	5,5,5,1,2	4.56 mg/kg	
Selenium	708	ng	Log	5.58	5.0	5,5,5,1,2	4.54 mg/kg	
Lead	3.88 [c)]	ng	Log	5.58	5.0	5,5,5,1,2	0.0332 mg/kg	
Tyre and brake wear								
Particulates	77.7	mg	Log	2.51	2.0	5,5,5,1,2	0.0777 g/km	3-1

[a)] The value is multiplied with 2 – Equation (19). [c)] The value is multiplied with 0.75 - Equation (20).

Weighted transport distances

The assumed transport distances in this thesis were based on actual treatment plants in Europe. Although transport distances vary across regions, the assumed values generally agree with other European LCA studies (Table 9.14). Still, to model a worst-case, a sensitivity analysis on the transport distances is performed in Chapter 10.2 as well. This also considered a possibly higher water content in DFG of 25%.

Table 9.14: Modelled transport distances to different types of treatment plants in Europe

References	Central plant	Metal recycling	Incineration	Landfill
This thesis	500	250	20	20
Shonfield, 2008	50-100	-	50	20
Jenseit et al. 2003	400-600	-	50	35
Laner et al., 2015	-	250	20	20

Vehicle type

In this thesis a transport vehicle with a capacity of 3.5-7.5 tonnes was assumed. However, apart from Fiore et al. (2019) who use a similar vehicle, other LCA studies typically considered much larger vehicles. For example, Shonfield (2008) modelled a 28-32 tonnes vehicle while Rossi et al. (2015) modelled a 20-28 tonnes lorry. Still, as larger vehicles are presumably more efficient, the current modelling represents a worst case. Given the relatively low impact of the transport process in general, changes to the vehicle type are not further investigated.

Big bags vs container

During this research project, DFG was collected in big bags. However, this is not only resourceful but also a challenge for some processes. For example, the incineration process does not accept waste in big bags because a time-consuming removal of the big bag cannot take place. However, such a pre-treatment is necessary because without it a full combustion could not take place. Therefore, the use of containers instead of big bags was suggested. Still, as this would increase the transport weight, further analysis on its effects would be needed.

9.4 Life Cycle Inventory summary

In Chapter 5-9 the process specific LCI models were built. Experimental processes served as data source for the waste composition and to better understand the technical constraints for processing DFG. To allow for a meaningful comparison of different scenarios, scale-up considerations for the experiments were discussed and modelled. The life cycle inventory was then built upon the most suitable technology choice considering typical separation efficiencies for the waste composition. The most critical modelling aspects were further discussed to highlight areas of uncertainty. Based on the established LCIs the next chapter is built.

10 Life Cycle Impact Assessment

10.1 Scope of the Life Cycle Impact Assessment

The life cycle impact assessment (LCIA) translates the life cycle inventories (Chapter 5-9) into potential environmental impacts for the selected impact categories. For ease of reading, the previously selected impact categories (Chapter 4) are once more summarised in Table 10.1.

Table 10.1: Selected impact categories

CC – Climate change	EPfw – Freshwater eutrophication	ETt – Terrestrial ecotoxicity
APt – Acidification	EPm – Marine eutrophication	ETfw – Freshwater ecotoxicity
HT – Human toxicity	POF – Photochemical oxidant formation	ETm – Marine ecotoxicity
WD – Water depletion	MD – Metal depletion	FD- Fossil depletion

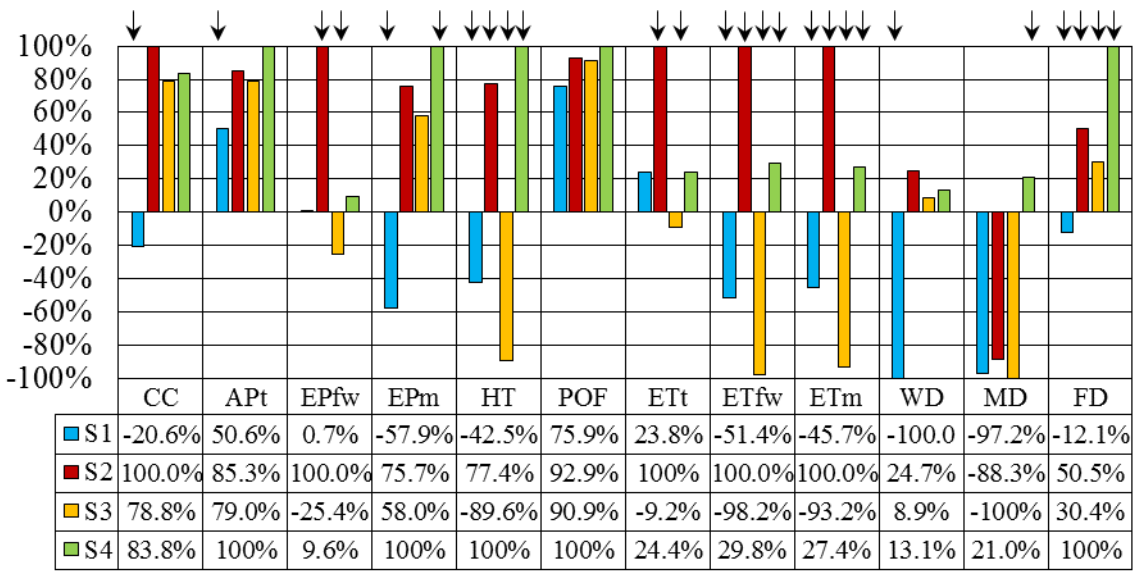
In this chapter, the potential environmental impacts are further analysed. First, an impact contribution analysis is conducted to compare the potential environmental impacts of the modelled mechanical recycling (S_1), chemical recycling (S_2), energy recovery (S_3) and disposal (S_4) scenario and to identify their most significant impact contributions. This is followed by a sensitivity analysis that investigates the robustness of selected input parameters and ultimately, an uncertainty analysis is carried out to examine the overall uncertainty of the results.

10.2 Impact contribution analysis

10.1.1 Relative scenario contributions

An overview of the total potential environmental impacts for each scenario is given in Figure 10.1. More specifically, the relative impact contributions of each scenario based on the highest absolute impact contributor within each impact category are shown. For example, the overall climate change results (S_1 = -559 CO_2-eq, S_2=2714 CO_2-eq, S_3 =2140 CO_2-eq, S_4=2274 kg CO_2-eq) were compared to scenario two and thus expressed as -20.6%, 100%, 78.8% and 83.8% for scenario 1-4 respectively (Figure 10.1). Although this presentation may appear complex first, it allows for all impact categories to be represented in a joint graph. Negative values indicate that production credits (i.e. heat production of the incineration process) dominate the impact score.

Focusing on the lowest scores in Figure 10.1, it becomes evident that the mechanical recycling and energy recovery scenario outperform the chemical recycling and disposal scenario across all selected impact categories. A more detailed analysis is provided below.



	CC	APt	EPfw	EPm	HT	POF	ETt	ETfw	ETm	WD	MD	FD
S1	-20.6%	50.6%	0.7%	-57.9%	-42.5%	75.9%	23.8%	-51.4%	-45.7%	-100.0	-97.2%	-12.1%
S2	100.0%	85.3%	100.0%	75.7%	77.4%	92.9%	100%	100.0%	100.0%	24.7%	-88.3%	50.5%
S3	78.8%	79.0%	-25.4%	58.0%	-89.6%	90.9%	-9.2%	-98.2%	-93.2%	8.9%	-100%	30.4%
S4	83.8%	100%	9.6%	100%	100%	100%	24.4%	29.8%	27.4%	13.1%	21.0%	100%

S1-Mechanical Recycling, S2-Chemical Recycling, S3-Energy Recovery, S4-Disposal

↓ Robust ranking result (Difference to other scenarios > 20%)

CC-Climate Change, APt-Terrestrial acidification, EPfw-Freshwater eutrophication, EPm-Marine eutrophication, HT-Human toxicity, POF-Photochemical oxidant formation, ETt-Terrestrial ecotoxicity, ETfw-Freshwater ecotoxicity, ETm-Marine ecotoxicity, WD-Water depletion, MD-Metal depletion, FD-Fossil depletion

Figure 10.1: Characterisation results – Scenario comparison

The mechanical recycling ranks first within CC, APt, EPm, POF, WD and FD and second within the remaining impact categories. Particularly for CC, APt, EPm and HT as well as for ETfw, ETm, WD and FD the ranking is very robust, because the impact contribution differs by at least 20% from the other scenarios (Figure 10.1).

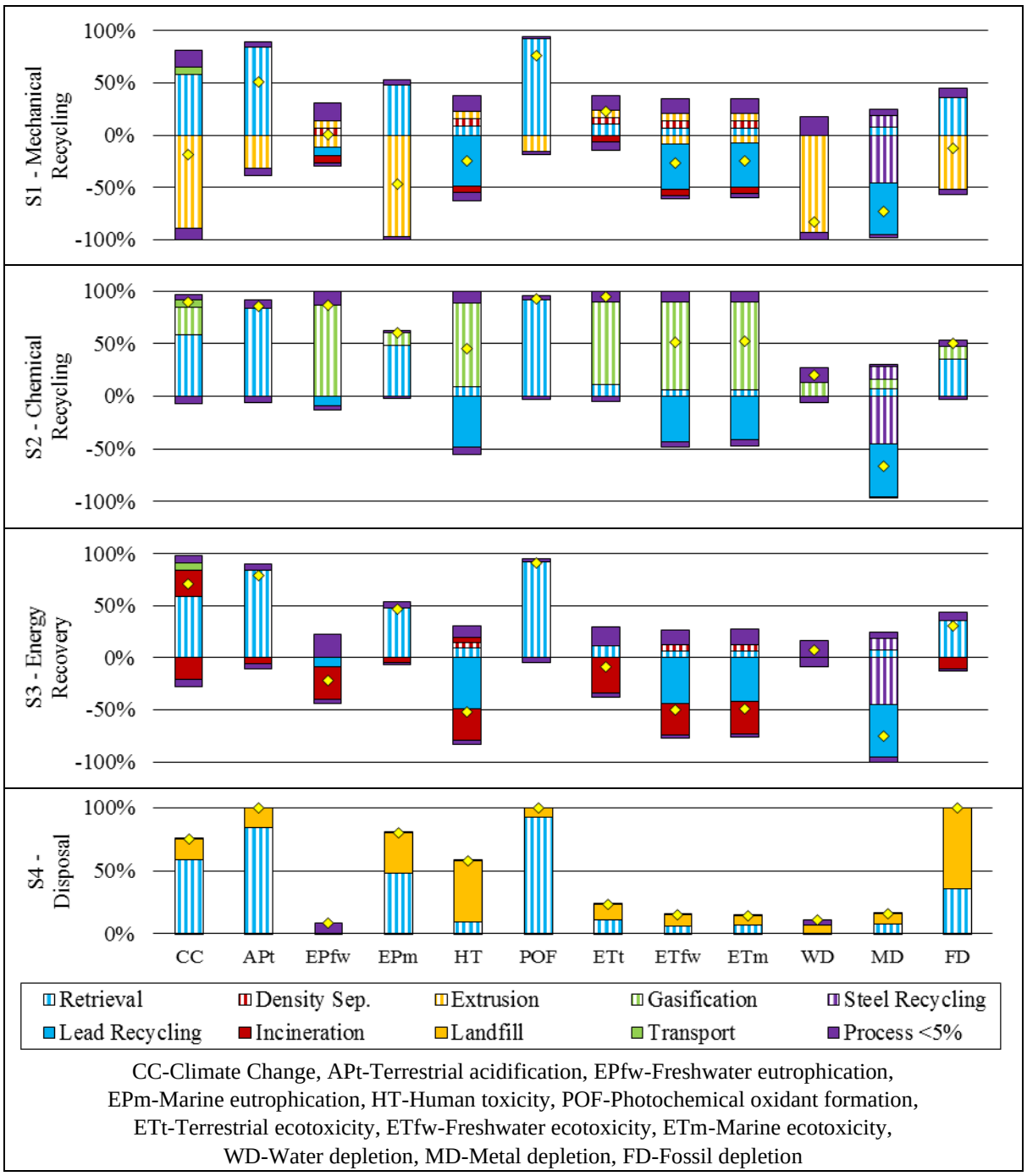
The energy recovery achieves the first rank for EPfw, HT, ETt, ETfw, ETm and MD and the second rank for the remaining impact categories. Robust ranking results that differ by at least 20% from other scenarios appear in EPfw, HT, ETt, ETfw, ETm and FD (Figure 10.1).

The chemical recycling scores the last rank for CC, EPfw, ETt, ETfw, ETm and WD, and the second last rank for the other impact categories. EPfw, HT, ETt, ETfw, ETm and FD are the impact categories with a robust ranking result that differs by at least 20% from the other scenarios (Figure 10.1).

The disposal scenario reaches the last rank for APt, EPm, HT, POF, MD and FD and the second last rank for the remaining impact categories. The most robust ranking results, with impact contributions that vary by at least 20% from the other scenarios, occur within EPm, HT, ETfw, ETm, MD and FD (Figure 10.1).

10.1.2 Relative process contributions

The relative impact contributions of the scenarios and their underlying processes are displayed in Figure 10.2-10.4. Figure 10.2 and Figure 10.3 follow a top down approach whereas the impact contributions in Figure 10.4 are displayed bottom-up. The former approach allows to trace back significant contributions while the latter approach enables to assess the joint impact contributions from underlying processes like the electricity production. Processes that contribute less than 5% to a specific impact category were grouped together.



CC-Climate Change, APt-Terrestrial acidification, EPfw-Freshwater eutrophication, EPm-Marine eutrophication, HT-Human toxicity, POF-Photochemical oxidant formation, ETt-Terrestrial ecotoxicity, ETfw-Freshwater ecotoxicity, ETm-Marine ecotoxicity, WD-Water depletion, MD-Metal depletion, FD-Fossil depletion

Figure 10.2: Relative impact contributions – Top down scenario view

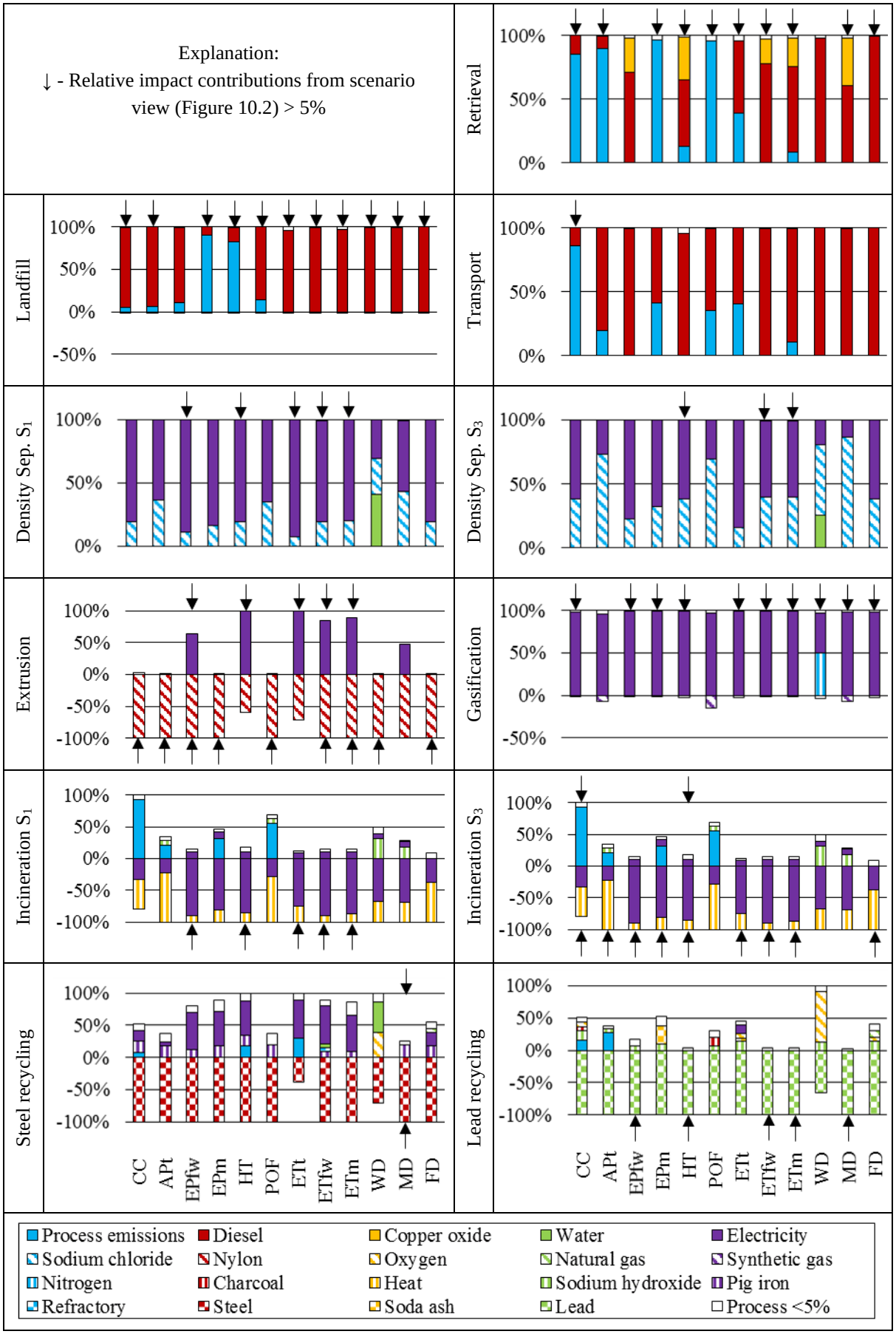


Figure 10.3: Relative impact contribution – Top down process view

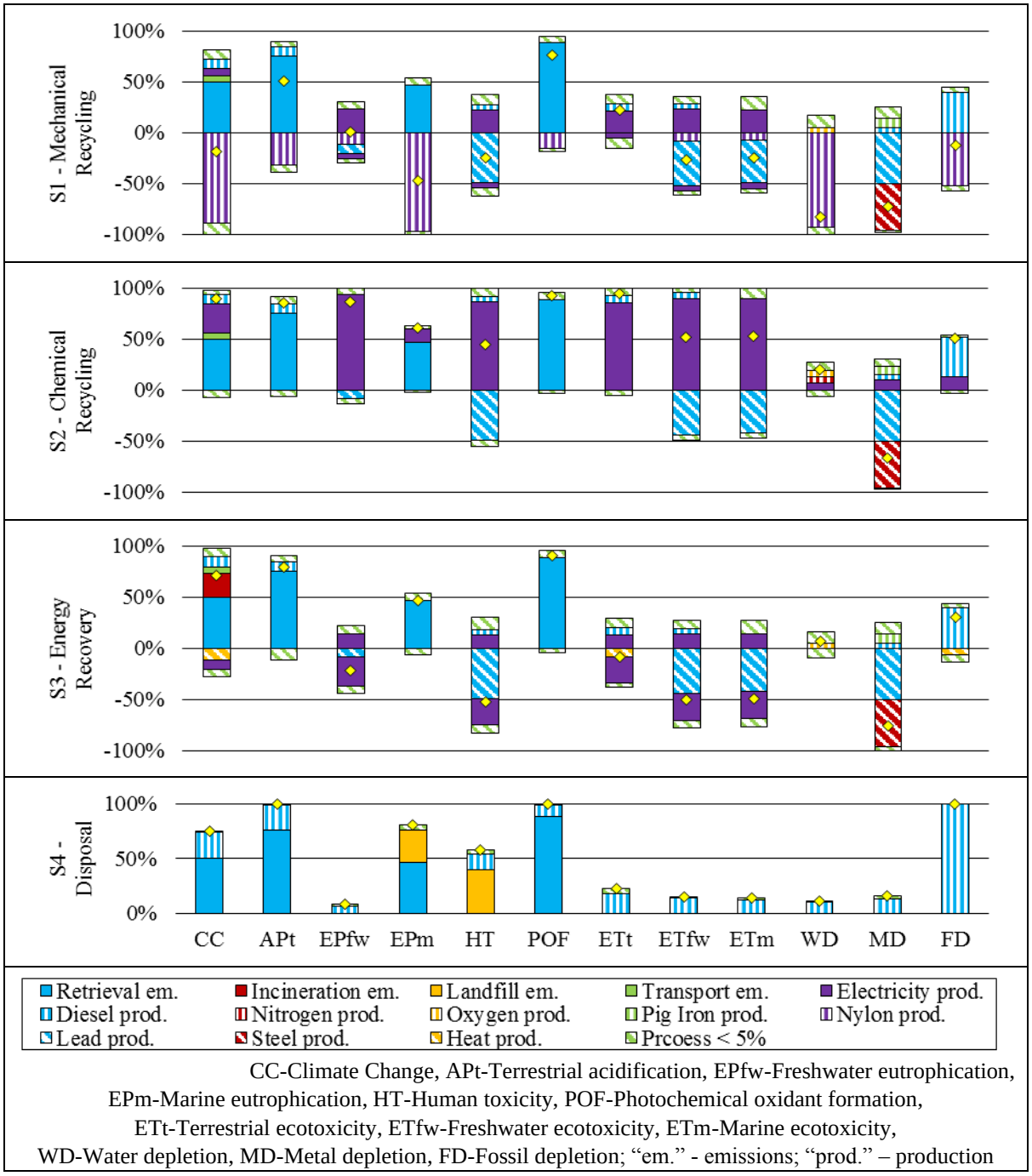


Figure 10.4: Relative impact contributions – bottom up scenario view

The retrieval scores the same relative impact contributions in all scenarios (Figure 10.2), because it is carried out independently from the waste treatment pathway. It contributed most to CC, APt, EPm, POF and FD (Figure 10.2), mainly due to its preceding diesel production and process emissions (Figure 10.3, Figure 10.4).

The impact contributions from the sorting, shredding, washing and drying process derive predominantly from the electricity production (Appendix C). As the processes contributed less than 5% to any of the impact categories, they are not displayed in Figure 10.2-10.4.

The impact contributions from the steel and lead recycling are the same in scenario 1-3 (Figure 10.2). While the steel recycling is particularly beneficial to MD (Figure 10.2), the lead recycling leads to additional benefits to EPfw, HT, ETfw and ETm (Figure 10.2) from an avoided steel or lead production (Figure 10.3, Figure 10.4). The steel recycling also causes notable contributions to MD from pig iron (Figure 10.4), while the joint oxygen demand from the steel and lead recycling (Figure 10.3) is notable within WD (Figure 10.4).

The impact contribution from the transport process mainly derives from direct emissions and the diesel production (Figure 10.3). However, its overall impact is relatively small and only visible within scenario 1-3 as part of CC (Figure 10.2).

The density separation is modelled as a two-stage process in scenario 1 and as a one-stage process in scenario 3. While both process variations exceed the 5% threshold to HT, ETfw and ETm, the two-stage process can also be noted in EPfw and ETt (Figure 10.2). As Figure 10.3 suggests, the impact contribution within the density separation is mainly linked to the electricity production.

The extrusion process significantly contributes to all impact categories apart from MD (Figure 10.2). For CC, APt, EPm, POF, WD and FD the avoided nylon production dominates the impact contribution (Figure 10.3, Figure 10.4). Yet, within EPfw, ETfw and ETm those credits are significantly reduced by contributions from the electricity production (Figure 10.3). In HT and ETt the electricity production dominates the impact score (Figure 10.3, Figure 10.4).

The gasification process is displayed across all impact categories except for POF and APt (Figure 10.2). While gasification impacts mainly derive from the electricity production, the contribution within WD is almost equally shared between the electricity and nitrogen production (Figure 10.3; Figure 10.4).

The incineration process appears in scenario 1 and 3. In scenario 1 credits are visible for EPfw, HT, ETt, ETfw and ETm while in scenario credits also occur in CC, APt, EPm and FD (Figure 10.2). In CC and FD the avoided heat production dominates the credits, whereas in other impact categories the avoided electricity production contributes most (Figure 10.3). Process emissions and the electricity production also contribute notably to CC and HT in scenario 3 (Figure 10.3).

Apart from EPfw, the landfill process is shown in all impact categories (Figure 10.2). For EPm and HT the main contributions derive from landfill emissions, whereas the remaining impact categories are dominated by the diesel production (Figure 10.3, Figure 10.4).

10.1.3 Absolute process contributions

Figure 10.5 and Figure 10.6 display the absolute impact contributions of the scenarios using a top down and bottom up approach. As before, process contributions below 5% are grouped.

Climate change

The largest absolute contributions to CC are attributed to the retrieval, gasification and incineration as well as the landfill and transport process (Table 10.2, Figure 10.5). At the same time, the extrusion and incineration process achieve significant credits as well (Table 10.2).

Table 10.2: Key contributions to climate change in [kg CO_2-eq / FU]

Top down process contributions					
	Process	S_1	S_2	S_3	S_4
+	Incineration	-		779	
	Retrieval	1771	1771	1771	1771
	Transport	209	209	209	-
	Landfill				496
	Gasification		806		
	Rest <5%	487	156	209	7.67
-	Extrusion	2693			
	Incineration	-		615	
	Rest <5%	334	228	213	0.467
	SUM	-559	2714	2140	2274

Bottom up process contributions					
	Process	S_1	S_2	S_3	S_4
[1)]	Incineration	-		719	
	Retrieval	1509	1509	1509	1509
	Transport	179	179	179	-
[2)]	Diesel	287	287	287	726
	Electricity	219	863	-	-
	Rest <5%	273	103	275	39.2
[3)]	Nylon	2693			
	Heat	-		354	-
	Electricity	-		261	-
	Rest <5%	334	228	213	0.467
	SUM	-559	2714	2140	2274

[1)] Process emissions. [2)] Material/energy production. [3)] Avoided material/energy production.

The credits from the extrusion process represent an avoided nylon production (Figure 10.3, Table 10.2), whereas the credits from the incineration process are split into an avoided heat and electricity production (Figure 10.3, Table 10.2). The incineration, retrieval and transport produced notable emissions (Figure 10.3, Table 10.2), while the latter two also required a significant diesel production (Table 10.2). The higher value for the diesel production in scenario 4 (Table 10.2) is linked to the landfill process (Figure 10.3). The high electricity production in scenario 2 was mainly caused by the gasification process. In scenario 1 several smaller processes like the extrusion, density separation and shredding jointly led to significant impact contributions from the electricity production (Table 10.2).

Acidification potential

The processes with the highest contribution to APt are the retrieval, extrusion and landfill as well as the incineration (Figure 10.5).

The credit from the extrusion represents an avoided nylon production (Figure 10.6), while the credit from the incineration process is split into an avoided heat and electricity production

(Figure 10.3). However, as the avoided heat and electricity production contributed less than 5% to the overall impact, they do not appear separately in Figure 10.6. The retrieval did not only generate direct emissions, but also contributed to the overall diesel production (Table 10.3). The higher impact from the diesel production in scenario 4 (Table 10.3) is attributed to the landfill process (Figure 10.3).

Table 10.3: Key contributions to terrestrial acidification in [kg SO_2-eq / FU]

Top down process contributions					
	Process	S_1	S_2	S_3	S_4
+	Retrieval	24.3	24.3	24.3	24.3
	Landfill				4.58
	Rest <5%	1.53	2.14	1.71	0.01
-	Extrusion	9.20			
	Incineration	-		1.49	
	Rest <5%	1.99	1.78	1.70	<<
	SUM	14.6	24.7	22.8	28.9

Bottom up process contributions					
	Process	S_1	S_2	S_3	S_4
1)	Retrieval	21.9	21.9	21.9	21.9
2)	Diesel	2.64	2.64	2.64	6.68
	Rest <5%	1.33	1.94	1.51	0.358
3)	Nylon	9.20			
	Rest <5%	1.99	1.78	3.19	<<
	SUM	14.6	24.7	22.8	28.9

[1)] Process emissions. [2)] Material/energy production. [3)] Avoided material/energy production.

Freshwater eutrophication

The key contributions to EPfw derive from the gasification, incineration and extrusion as well as the lead recycling and density separation (Table 10.4, Figure 10.5).

Table 10.4: Key contributions to freshwater eutrophication in [kg P-eq / FU]

Top down process contributions					
	Process	S_1	S_2	S_3	S_4
+	Density Sep.	0.086		-	
	Extrusion	0.096			
	Gasification		1.16		
	Rest <5%	0.227	0.173	0.294	0.111
-	Extrusion	0.151			
	Lead recycling	0.116	0.116	0.116	
	Incineration	0.082		0.420	
	Rest <5%	0.051	0.062	0.051	0.001
	SUM	0.008	1.15	-0.292	0.110

Bottom up process contributions					
	Process	S_1	S_2	S_3	S_4
1)	Diesel	-	-	-	0.090
	Electricity	0.315	1.25	0.191	-
	Rest <5%	0.094	0.084	0.103	0.021
2)	Nylon	0.151			
	Lead	0.116	0.116	0.116	
	Electricity	0.074		0.376	
	Rest <5%	0.060	0.062	0.095	0.001
	SUM	0.008	1.15	-0.292	0.110

[1)] Material/energy production. [2)] Avoided material/energy production.

The credits from the lead recycling and extrusion represent an avoided lead and nylon production (Table 10.4), while the credits from the incineration process are divided into approximately 90% electricity and 10% heat production (Figure 10.3). However, from those, only the avoided electricity production exceeds the 5% threshold (Table 10.4). The impact from the electricity production is notable in scenario 1-3 (Table 10.4), while the diesel production only appears in scenario 4 (Table 10.4).

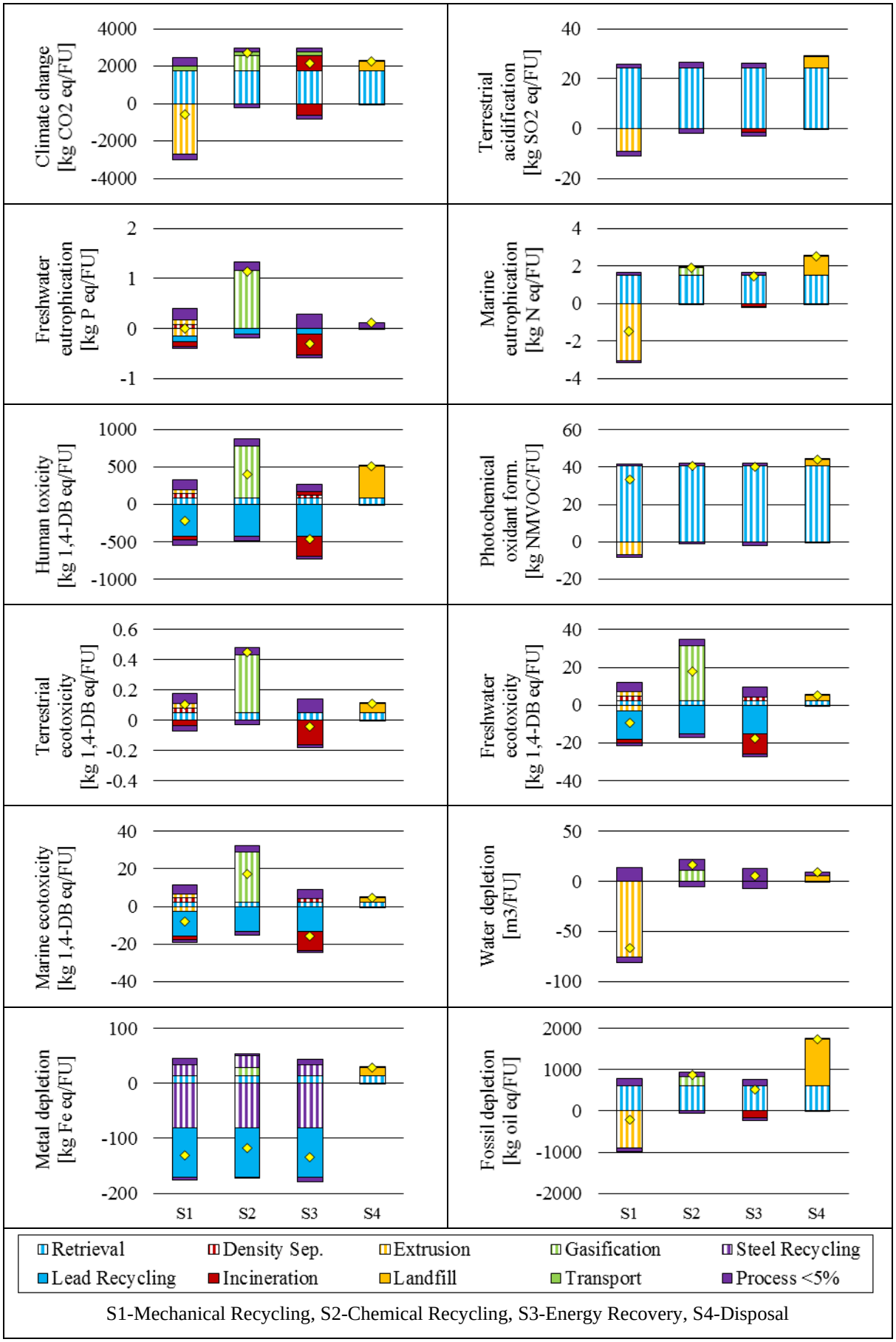
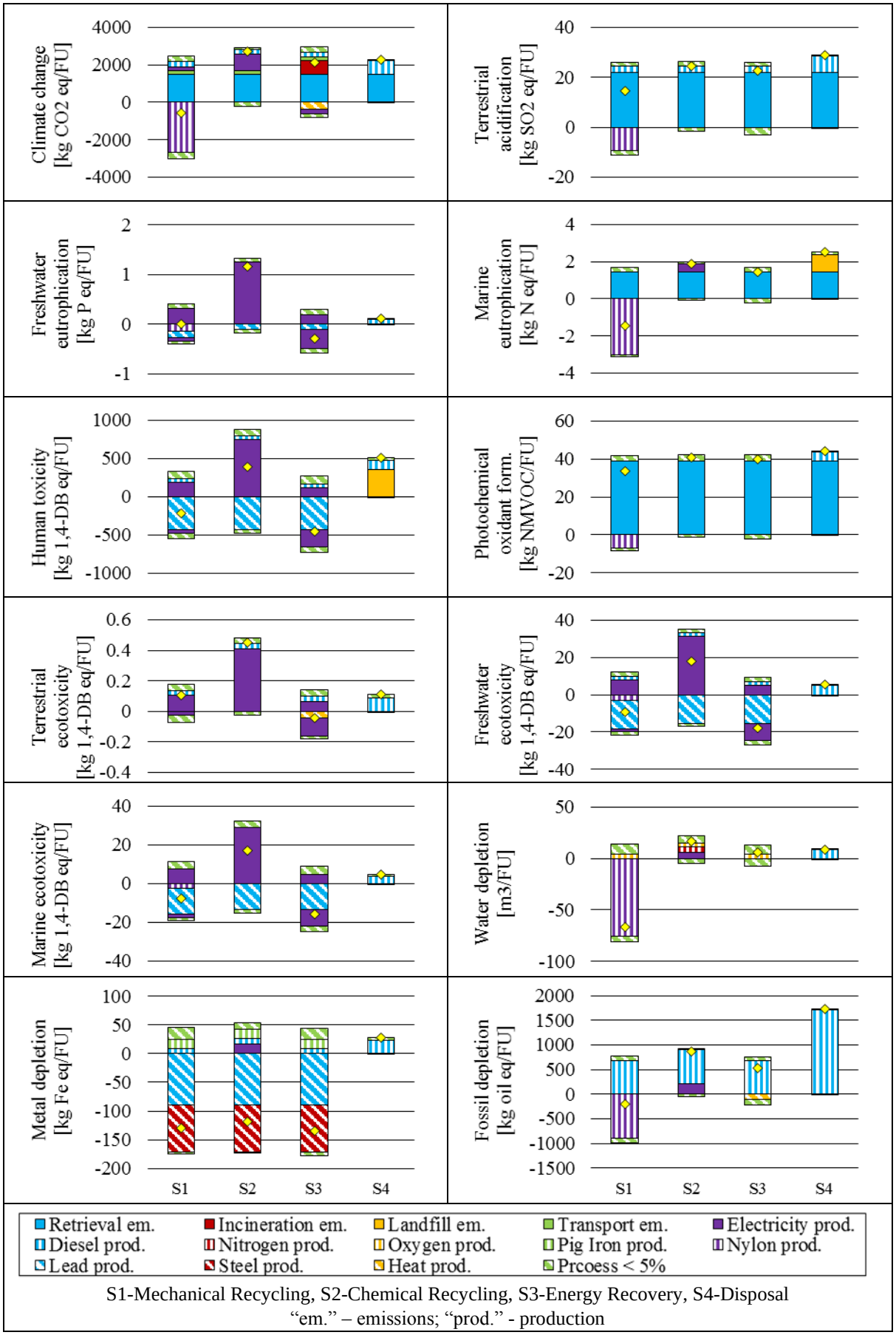


Figure 10.5: Absolute impact contributions – top down scenario view



S1-Mechanical Recycling, S2-Chemical Recycling, S3-Energy Recovery, S4-Disposal
"em." – emissions; "prod." - production

Figure 10.6: Absolute impact contributions – bottom up scenario view

Marine eutrophication

The main contributions to EPm derive from the extrusion, retrieval and landfill as well as the gasification and incineration process (Table 10.5, Figure 10.5).

The credits from the extrusion embody an avoided nylon production (Figure 10.6), whereas the credits from the incineration are split into an avoided electricity and heat production (Figure 10.3). Again, the avoided heat and electricity production were too small to be individually displayed in Figure 10.6. The retrieval and landfill process caused significant process emissions, while the gasification mainly contributed to the electricity production in scenario 2 (Table 10.5).

Table 10.5: Key contributions to marine eutrophication in [kg N-eq / FU]

Top down process contributions					
	Process	S_1	S_2	S_3	S_4
+	Retrieval	1.51	1.51	1.51	1.51
	Landfill				1.01
	Gasification		0.390		
	Rest <5%	0.161	0.068	0.163	<<
-	Extrusion	3.05			
	Incineration	-		0.157	
	Rest <5%	0.087	0.062	0.056	<<
	SUM	-1.46	1.91	1.46	2.52

Bottom up process contributions					
	Process	S_1	S_2	S_3	S_4
1)	Retrieval	1.46	1.46	1.46	1.46
	Landfill				0.911
2)	Electricity	-	0.419	-	-
	Rest <5%	0.217	0.095	0.220	0.153
3)	Nylon	3.05			
	Rest <5%	0.087	0.062	0.213	<<
	SUM	-1.46	1.91	1.46	2.52

[1)] Process emissions. [2)] Material/energy production. [3)] Avoided material/energy production.

Human toxicity

The key process contributions to HT are the gasification, landfill and retrieval as well as the density separation and extrusion (Table 10.6, Figure 10.5). The incineration and lead recycling lead to relevant negative contributions (Table 10.6).

Table 10.6: Key contributions to human toxicity in [kg 1,4-DB-eq / FU]

Top down process contributions					
	Process	S_1	S_2	S_3	S_4
+	Landfill				429
	Retrieval	80.2	80.2	80.2	80.2
	Incineration	-		47.7	
	Density Sep.	61.3		46.6	
	Extrusion	58.4			
	Gasification		700		
	Rest <5%	132	96.2	96.2	0.188
-	Lead recycling	428	428	428	
	Incineration	52.0		266	
	Rest <5%	68.3	54.7	33.8	0.332
	SUM	-216	394	-456	510

Bottom up process contributions					
	Process	S_1	S_2	S_3	S_4
1)	Landfill				353
2)	Diesel	46.9	46.9	46.9	119
	Electricity	191	754	116	-
	Rest <5%	93.8	76.1	108	38.5
3)	Lead	428	428	428	
	Electricity	-		228	
	Rest <5%	120	54.7	71.9	0.332
	SUM	-216	394	-456	510

[1)] Process emissions. [2)] Material/energy production. [3)] Avoided material/energy production.

The credits for the lead recycling derive from an avoided lead production (Figure 10.3, Table 10.6), whereas the credits from the incineration process are mainly attributed to an avoided

electricity production (Figure 10.3, Table 10.6). The landfill process causes significant direct emissions (Table 10.6), while contributing to the otherwise retrieval related diesel production (Table 10.6). The incineration, density separation and extrusion as well as the gasification significantly contribute to the electricity production (Table 10.6).

Photochemical oxidant formation

The key contributions to POF are attributed to the retrieval, extrusion and landfill process (Table 10.7, Figure 10.5). The extrusion achieves negative impact contributions that represent an avoided nylon production (Table 10.7). Although the retrieval releases most emissions directly (Table 10.7), in combination with the landfill process, it also causes a significant impact from the diesel production (Table 10.7).

Table 10.7: Key contributions to photochemical oxidant form. in [kg NMVOC-eq / FU]

Top down process contributions					
	Process	S_1	S_2	S_3	S_4
+	Retrieval	40.6	40.6	40.6	40.6
	Landfill				3.41
	Rest <5%	1.09	1.53	1.44	0.011
-	Extrusion	7.02			
	Rest <5%	1.27	1.22	2.00	0.001
	SUM	33.4	40.9	40.0	44.0

Bottom up process contributions					
	Process	S_1	S_2	S_3	S_4
1)	Retrieval	38.9	38.9	38.9	38.9
2)	Diesel	-	-	-	4.56
	Rest <5%	2.76	3.21	3.11	0.539
3)	Nylon	-7.02			
	Rest <5%	1.27	1.22	2.00	0.001
	SUM	33.4	40.9	40.0	44.0

[1)] Process emissions. [2)] Material/energy production. [3)] Avoided material/energy production.

Terrestrial ecotoxicity

The key process contributions to ETt derive from the gasification, landfill and retrieval as well as the extrusion and density separation (Table 10.8, Figure 10.5). Significant negative contributions are attributed to the incineration process (Table 10.8).

Table 10.8: Key contributions to Terrestrial Ecotoxicity in [kg 1,4-DB-eq / FU]

Top down process contributions					
	Process	S_1	S_2	S_3	S_4
+	Retrieval	0.053	0.053	0.053	0.053
	Landfill				0.057
	Density Sep.	0.026		-	
	Extrusion	0.032			
	Gasification		0.378		
	Rest <5%	0.068	0.048	0.086	<<
-	Incineration	0.032		0.164	
	Rest <5%	0.040	0.026	0.017	<<
	SUM	0.108	0.453	-0.042	0.111

Bottom up process contributions					
	Process	S_1	S_2	S_3	S_4
2)	Diesel	0.034	0.034	0.034	0.085
	Electricity	0.103	0.408	0.063	-
	Rest <5%	0.042	0.037	0.043	0.026
3)	Electricity	0.024		0.123	
	Heat	-		0.041	
	Rest <5%	0.048	0.026	0.017	<<
	SUM	0.108	0.453	-0.042	0.111

[1)] Process emissions. [2)] Material/energy production. [3)] Avoided material/energy production.

The credits from the incineration process are divided into an avoided electricity and heat production (Figure 10.3, Table 10.8). However, the avoided heat production in scenario 1 does not

exceed the 5% threshold. The retrieval and landfill contributed notably to the diesel production (Table 10.8), whereas the energy intensive processes such as the gasification, extrusion and density separation added to the electricity production (Table 10.8).

Freshwater ecotoxicity

The main contributions to ETfw come from the gasification, landfill and extrusion as well as the density separation and retrieval process (Table 10.9, Figure 10.5). The lead recycling, incineration and extrusion lead to significant negative contributions (Table 10.9).

Table 10.9: Key contributions to Freshwater Ecotoxicity in [kg 1,4-DB-eq / FU]

Top down process contributions					
	Process	S_1	S_2	S_3	S_4
+	Retrieval	2.21	2.21	2.21	2.21
	Landfill				3.15
	Density Sep.	2.61		1.99	
	Extrusion	2.44			
	Gasification		29.2		
	Rest <5%	4.98	3.57	5.22	0.007
-	Extrusion	2.89			
	Lead recycling	15.3	15.3	15.3	
	Incineration	-		10.5	
	Rest <5%	3.33	1.75	1.26	0.014
	SUM	-9.23	18.0	-17.6	5.35

Bottom up process contributions					
	Process	S_1	S_2	S_3	S_4
[2)]	Diesel	1.91	1.91	1.91	4.83
	Electricity	7.96	31.5	4.83	-
	Rest <5%	2.37	1.61	2.68	0.531
[3)]	Nylon	2.89		0.123	
	Lead	15.3	15.3	15.3	
	Electricity	1.86		9.50	
	Rest <5%	1.47	1.75	2.31	0.014
	SUM	-9.23	18.0	-17.6	5.35

[1)] Process emissions. [2)] Material/energy production. [3)] Avoided material/energy production.

The credits for the incineration process mainly derive from an avoided electricity production (Figure 10.3, Table 10.9), whereas the credits from the lead recycling and extrusion are fully attributed to the avoided lead and nylon production (Table 10.9). The landfill and retrieval mainly contribute to the diesel production (Table 10.9) while the density separation, extrusion and gasification mainly added to the scenario specific electricity production (Table 10.9).

Marine ecotoxicity

The major process contributions to ETm come from the gasification, landfill and extrusion as well as the density separation and retrieval (Table 10.10, Figure 10.5). The lead recycling, incineration and extrusion achieve notable credits (Table 10.10).

The credits to ETm derive primarily from an avoided electricity production (Table 9.11) or represent an avoided nylon and lead production (Table 10.10). The retrieval and landfill process require a notable diesel production, whereas the other processes contributed primarily to the electricity production (Table 10.10).

Table 10.10: Key contributions to Marine Ecotoxicity in [kg 1,4-DB-eq / FU]

Top down process contributions					
	Process	S_1	S_2	S_3	S_4
+	Retrieval	2.07	2.07	2.07	2.07
	Landfill				2.60
	Density Sep.	2.41		1.85	
	Extrusion	2.24			
	Gasification		26.8		
	Rest <5%	4.64	3.32	4.86	0.007
-	Extrusion	2.51			
	Lead recycling	13.5	13.5	13.5	
	Incineration	1.95		10.0	
	Rest <5%	1.25	1.70	1.25	0.013
	SUM	-7.80	17.0	-15.9	4.66

Bottom up process contributions					
	Process	S1	S3	S4	S5
1)	Diesel	-	-	-	3.92
	Electricity	7.30	28.9	4.43	-
	Rest <5%	4.06	3.33	4.35	0.751
2)	Nylon	2.51		0.123	
	Lead	13.5	13.5	13.5	
	Electricity	1.71		8.72	
	Rest <5%	1.49	1.70	2.49	0.013
	SUM	-7.80	17.0	-15.9	4.66

[1)] Material/energy production. [2)] Avoided material/energy production.

Water depletion

The most significant processes to WD are the extrusion, gasification and landfill (Figure 10.6). From those, only the extrusion leads to negative impact contributions (Table 10.11).

Table 10.11: Key contributions to Water depletion in [m3 / FU]

Top down process contributions					
	Process	S_1	S_2	S_3	S_4
	Gasification		10.7		
	Landfill				5.61
	Rest <5%	14.0	11.0	13.3	3.17
-	Extrusion	75.4			
	Rest <5%	5.37	5.26	7.37	0.003
	SUM	-66.8	16.5	5.91	8.77

Bottom up process contributions					
	Process	S_1	S_2	S_3	S_4
1)	Electricity	-	5.50	-	-
	Nitrogen	-	5.43	-	
	Diesel	-	-	-	8.69
	Oxygen	4.31	4.31	4.31	
	Rest <5%	9.68	6.48	8.98	0.082
2)	Nylon	75.4			
	Rest <5%	5.37	5.26	7.37	0.003
	SUM	-66.8	16.5	5.91	8.77

[1)] Material/energy production. [2)] Avoided material/energy production.

The credits from the extrusion process stem from an avoided nylon production (Table 10.11). The gasification process mainly requires water for the electricity and the nitrogen production (Figure 10.3), while the landfill process together with the retrieval require water for the diesel production. The oxygen demand of the steel and lead recycling is also linked to a notable water depletion.

Metal depletion

The main process contributions to MD are the lead recycling, steel recycling and gasification as well as the landfill and retrieval (Table 10.12; Figure 10.5). From those, the steel and lead recycling cause significant negative impact contributions (Table 10.12).

Those credits from the steel and lead recycling represent an avoided steel and lead production (Table 10.12). The impacts from the steel recycling are mainly linked to the pig iron production

(Figure 10.3, Table 10.12), whereas the retrieval and landfill notably contributed to the diesel production (Table 10.12). The gasification process caused a substantial part of the overall electricity production (Table 10.12).

Table 10.12: Key contributions to Metal depletion in [kg Fe-eq / FU]

Top down process contributions					
	Process	S_1	S_2	S_3	S_4
+	Retrieval	13.3	13.3	13.3	13.3
	Landfill				14.8
	Steel recycling	20.5	20.5	20.5	
	Gasification		16.3		
	Rest <5%	11.1	3.45	10.6	0.035
-	Steel recycling	80.8	80.8	80.8	
	Lead recycling	89.9	89.9	89.9	
	Rest <5%	4.35	1.10	7.62	0.008
	SUM	-130	-118	-134	28.1

Bottom up process contributions					
	Process	S_1	S_2	S_3	S_4
1)	Diesel	9.05	9.05	9.05	22.9
	Pig iron	15.5	15.5	15.5	
	Electricity	-	17.3	-	-
	Rest <5%	20.3	11.6	19.8	5.20
2)	Steel	80.8	80.8	80.8	
	Lead	89.9	89.9	89.9	
	Rest <5%	4.35	1.10	7.62	0.008
	SUM	-130	-118	-134	28.1

[1)] Material/energy production. [2)] Avoided material/energy production.

Fossil depletion

The landfill, extrusion and retrieval as well as the gasification and incineration process contributed more than 5% to FD (Table 10.13, Figure 10.5). From those, the extrusion and incineration had negative impact contributions (Table 10.13).

Table 10.13: Key contributions to Fossil depletion in [kg oil-eq / FU]

Top down process contributions					
	Process	S_1	S_2	S_3	S_4
+	Retrieval	616	616	616	616
	Landfill				1111
	Gasification		206		
	Rest <5%	159	106	136	2.62
-	Extrusion	902			
	Incineration	-		177	
	Rest <5%	83.6	53.9	48.9	0.125
	SUM	-209	874	526	1730

Bottom up process contributions					
	Process	S_1	S_2	S_3	S_4
1)	Diesel	681	681	681	1724
	Electricity	-	221	-	-
	Rest <5%	94.7	26.8	71.0	6.45
2)	Nylon	902			
	Heat	-		111	
	Rest <5%	83.6	53.9	116	0.125
	SUM	-209	874	526	1730

[1)] Material/energy production. [2)] Avoided material/energy production.

The credit from the extrusion process represents an avoided nylon production (Table 10.13), whereas the credit from the incineration process is mainly linked to an avoided heat production (Table 10.13). The landfill and the retrieval process significantly contribute to the diesel production (Table 10.13), while the electricity intensive processes in particular the gasification caused most of the electricity production (Table 10.13).

10.3 Sensitivity analysis

The LCI is subject to various assumptions. To determine the robustness of the scenario ranking (Figure 10.1), a sensitivity analysis on the key assumptions was performed. This included the DFG composition, transport distances and market substitution factors as well as the underlying energy mix.

10.3.1 Setup

DFG composition

The baseline scenario assumes a waste composition of 50% gillnets and 50% trawl nets. Unlike trawl nets, gillnets contain lead which can cause higher toxic emissions. To investigate the difference, the collection and treatment of trawl and gillnets was modelled separately. For this, specific data about the process inputs (Table 10.14), the syngas output as well as the energy generation, lead and carbon emissions from the landfill process (Table 10.15) were adopted.

Table 10.14: Trawl and gillnet specific data (1)

Process	**Unit**	**Gillnets / trawl nets**				**References**
		S_1	**S_2**	**S_3**	**S_4**	
Sorting	kg	966 / 1000				Table 5.8
Shredding	kg	908 / 941				Table 5.14
1. Density Sep.	kg	893 / 926		893 / 926		Table 6.7
2. Density Sep.	kg	510 / 419				Table 6.8
Washing	kg	463 / 419				Table 6.14
Drying	kg	370 /308				Table 6.16
Extrusion	kg	370 / 308				Table 7.2
Gasification	kg		893 / 926			Table 7.7
Steel recycling	kg	72.9 / 74.5				Table 8.1
Lead Recycling	kg	135 / 0				Table 8.3
Incineration	kg	112 / 54.3		510 / 419		Table 9.1-9.2
Landfill	kg				966 / 1000	Table 5.4
Transport	km	554 / 542	552 / 541	552 /541	19.7 / 20.4	Table 9.11

Table 10.15: Trawl and gillnet specific data (2)

Process	**Flow**	**Name**	**Unit**	**Baseline**	**Trawl nets**	**Gillnets**	**References**
Gasification	Avoided Production	Syngas	m^3	0.418	0.351	0.486	Table 7.9
Landfill	Avoided production	Electricity	Wh	0.705	0.605	0.805	Table 9.10
		Heat	Wh	0.347	0.298	0.396	
	Emissions to air	Carbon dioxide	g	6.00	5.14	6.85	
		Methane	mg	866	743	989	
		Lead	mg	22.3	0	44.6	
	Emissions to water	TOC	mg	15.8	13.5	18.0	
		Lead	mg	15.9	0	31.8	

Transport

The baseline scenarios are based on a centralised logistic approach with Plastix Global as the key recycling facility. It assumes a dry material composition and an initial transport distance of 500 km (Chapter 9.3). For the sensitivity analysis a best and a worst case were established.

The worst-case accounts for the DFG wet weight and assumed Aquafil in Slovenia as the centralised recycling plant. For this, a water content of 25% and an initial transport distance of 1300 km were assumed. The best case represents a decentral logistic approach with a direct pre-treatment at the harbour. Apart from an initial transport distance of 0 km, a dry material composition is assumed. This resulted in the transport efforts in Table 10.16. For scenario 4 only the higher water content was added to the worst-case.

Table 10.16: Transport distance in [km] for the baseline, worst- and best-case

Cases	**Scenario 1**	**Scenario 2**	**Scenario 3**	**Scenario 4**
Baseline	548	546	547	20.1
Worst-case	1688	1686	1687	25.1
Best-case	165	277	55	20.1

Market substitution

The amount and quality of the waste treatment products can vary substantially, for which several best and worst cases were assumed. The baseline for the extrusion process already represents a best case as it assumes a nylon substitution of 100%. For the worst case a substitution ratio of 81% was assumed (Table 10.17).

Table 10.17: Product substitutions for the baseline, worst- and best-case

Process	**Avoided production**	**Unit**	**Baseline**	**Best case**	**Worst case**
Extrusion	Nylon pellets	kg	0.982	As baseline	0.795
		%	100		81.0
Gasification	Syngas	kg	0.418	1.31	As baseline
		%.	100	312.5	
Incineration S_1	Heat	kWh	2.86	5.57	-
		%	37.0	72.0	-
	Electricity	kWh	1.16	2.71	0.387
		%	15.0	35.0	5.0
Incineration S_3	Heat	kWh	2.61	5.08	-
		%	37.0	72.0	-
	Electricity	kWh	1.06	2.47	0.353
		%	15.0	35.0	5.0

The baseline model of the gasification process on the other hand represents a worst case. This is because the ancillary materials such as water were not added to the syngas quantity and because the quality of the syngas output is expected to be superior to the modelled syngas

(Chapter 7.2.4). As best case, a syngas output increase of 25% and a market substitution factor of 250% were adopted. This is based on the required water content for the gasification process and the approximately 2.5 times higher heating value of the produced syngas compared to standard market products. The resulting substitution factor of 312.5% was used to calculate the best case for the avoided syngas production in Table 10.17.

The incineration process is modelled based on an average European technology. Still, as varying thermal (3-72%) and electrical (5-35%) conversion efficiencies (Laner et al., 2015) can affect LCA results (i.e. Stamford, Greening and Azapagic, 2018), a best and a worst case for those parameters was assumed (Table 10.17). For the worst case, a thermal conversion efficiency of zero was adopted (Table 10.17) to represent an incineration plant that utilises electricity only. For the best case the highest conversion efficiencies for both the heat and electricity production were assumed.

Energy mix

The 2017 electricity mix for Germany (Table 10.18) was applied to the baseline scenarios. Yet, to reach the global warming goal set out in the Paris Agreement, the lignite and thus climate change intensive energy mix (Table C.16) is subject to change. Thus, a future energy mix with a higher share of renewable energy is proposed for the sensitivity analysis. While for similar considerations long-term changes until 2070 can be considered (Stamford and Azpagic, 2014), a more certain medium-term electricity mix for Germany 2030 is modelled here (Table 10.18). Germany was selected because most waste management processes took place there. The possible effect when choosing another country and thus electricity mix for the modelling is further evaluated as part of the discussion (Chapter 11.3) below.

Table 10.18: Electricity production mix

Name		Baseline		Energy 2030		References
		TWh	%	TWh	%	
Input						
	Water	20.1	3.7	20	3.4	Baseline - Frauenhofer, 2018 Energy 2030 - Agora Energiewende, 2017
	Biomass	44.7	8.2	30	5.2	
	Wind	106	19.3	250	43.1	
	Solar	39.4	7.2	70	12.1	
	Gas	49.1	9.0	70	12.1	
	Lignite	134	24.5	60	10.3	
	Hard coal	81.7	14.9	80	13.8	
	Nuclear	72.2	13.2	0	0.0	

10.3.2 Results

The results of the sensitivity analysis are provided in Figure 10.7. Compared to POF, APt, WD, FD, CC and EPm, the impact categories HT, ETm, MD, ETfw, EPfw and ETt are very sensitive as their results vary more widely from the baseline scenario (Figure 10.7).

Waste composition

The higher lead content in gillnets increased the credits from the lead recycling for scenario 1-3, while causing higher landfill emissions in scenario 4 (see HT, Figure 10.7). Thus, the chemical recycling (S2) and disposal (S4) swapped rankingsin ETfw and ETm (Figure 10.8).

In reverse, the lower lead content in trawl nets reduced the credits from the lead recycling in scenario 1-3, while causing lower emissions in scenario 4 (see HT, Figure 10.7). Consequently, the disposal scenario jumped from the last to the second place within HT, while changing the ranking positions with scenario 1 in EPfw, ETt, ETfw and ETm (Figure 10.8).

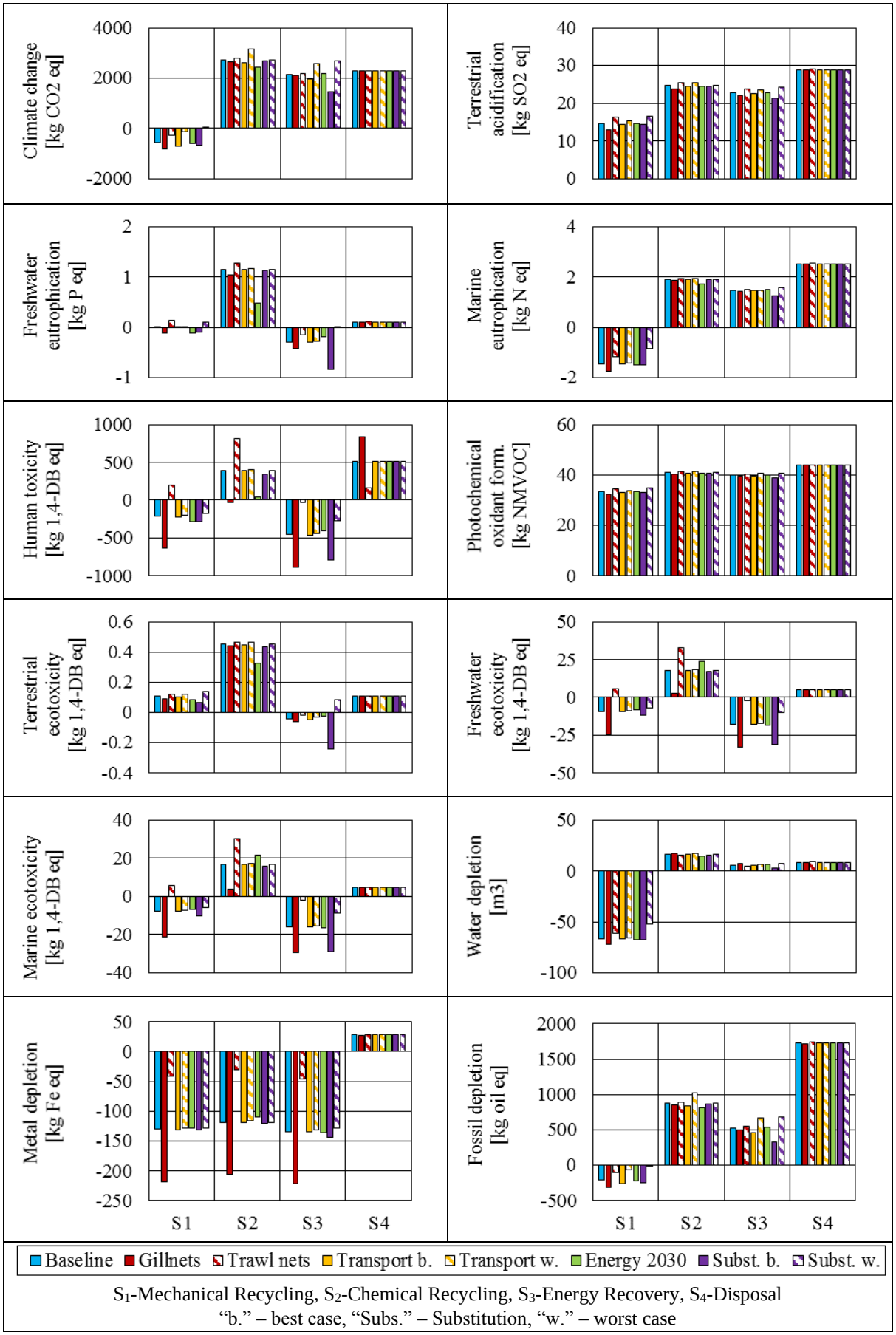
Transport

The investigated transport cases cause small fluctuations to the baseline within scenario 1-3, while the impact contribution from scenario 4 remains largely unchanged (Figure 10.7).The decentral logistic approach with shorter transport distances in scenario 1-3 leads to savings across all impact categories without affecting the ranking positions (Figure 10.7).

The central logistic approach based on Aquafil, increased the impact contributions for scenario 1-3. This allowed scenario 4 to swap ranking positions with scenario 1 and 3 in ETt and CC respectively (Figure 10.8). As the difference of the impact contribution was already small among the baseline scenarios (Figure 10.1), not too much attention should be given to those transport results.

Energy mix

The effect of a different energy mix can be best observed within the energy intensive chemical recycling scenario. This is particularly true for the impact categories ETt, ETfw, ETm, EPfw and HT where the changes to the baseline exceed 20% (Figure 10.7). Interestingly, ETfw, ETm and MD generate opposite results compared to the other impact categories (Figure 10.7). It implies, that the modelled future energy mix causes higher impacts within ETfw, ETm and MD while reducing the impacts within the other impact categories. This can be explained by the higher share of wind energy (Appendix C).



S_1-Mechanical Recycling, S_2-Chemical Recycling, S_3-Energy Recovery, S_4-Disposal
"b." – best case, "Subs." – Substitution, "w." – worst case

Figure 10.7: Sensitivity analysis results

In contrast to the other scenarios, scenario 3 generates more electricity than it consumes, causing impacts of opposite signs. In impact categories of lower future impacts such as HT, this means that the energy consuming scenarios improve their results, while the energy generating scenario 3 loses credits. Although the difference between the impact contributions of the scenarios shrinks, it does not cause changes to the scenario ranking.

Market substitution

The effects of the modelled market substitution factors become evident in CC, EPfw, EPm, HT, ETt, ETfw and ETm where impact contributions vary by more than 20% from the baseline (Figure 10.7). While the impact contributions remained the same for scenario 4, they decreased as part of the best-case modelling for the other scenarios. However, adopting a best case did not change the scenario ranking.

The worst-case modelling increased the net impact of scenario 1 and 3, causing changes to the ranking results in CC and ETt (Figure 10.8). Within MD, scenario 3 and 1 swapped positions. This is because the impact contribution in scenario 3 increased more rapidly than in scenario 1 due to its higher electricity generation. However, the initial gap between the two scenarios was very small (Figure 10.1), so that not too much focus should be paid on this result.

Ranking	Gillnets	Trawl nets		Transport, worst case		Substitution, worst case		
	ETfw, ETm	HT	EPfw,ETt ETfw, ETm	CC	ETt	CC	ETt	MD
1	S3	S3	S3	S1	S3	S1	S3	S3
2	S1	S1	S1	S3	S1	S3	S1	S1
3	S4	S2	S4	S4	S4	S4	S4	S2
4	S2	S4	S2	S2	S2	S2	S2	S4

Figure 10.8: Changes to the scenario ranking positions

10.4 Uncertainty analysis

The data points from the Monte Carlo analysis with 10,000 iterations were used to plot the scenario and impact category specific probability distributions (Figure 10.9) and corresponding 50%, 95% and 100% confidence intervals (Figure 10.10).

In all iterations, scenario 1-3 outperformed scenario 4 in MD, while scenario 3 surpassed scenario 2 in ETt (Table 10.19). Otherwise, the 100% confidence intervals overlapped (Figure 10.9), so that a clear winner could not be determined. Still, in at least 95% of the cases, scenario 1 ranked first in CC while outperforming scenario 4 in EPm and FD and scenario 2 additionally in ETt (Table 10.19). At the same level of confidence, scenario 3 ranked first in ETt (Table 10.19). It also surpassed scenario 2 and 4 in EPfw while beating scenario 4 in HT, ETfw and ETm (Table 10.19). Scenario 4 exceeded scenario 2 in ETt (Table 10.19).

Table 10.19: Significant ranking results

↓ outperforms →	CC				EPfw				EPm				HT, ETm				ETt				ETfw				MD				FD			
	Scenario				Scenario				Scenario				Scenario				Scenario				Scenario				Scenario				Scenario			
	1	2	3	4	1	2	3	4	1	2	3	4	1	2	3	4	1	2	3	4	1	2	3	4	1	2	3	4	1	2	3	4
S_1		x	x	x						x		x						x										x		x		x
S_2																												x				
S_3						x		x								x	x	x		x				x				x				
S_4																		x														

"x" – 100% confidence, "x" – 95% confidence

Uncertainty ranges and probability distributions

A short bandwidth is an indicator of robust and reliable results whereas a long bandwidth represents higher uncertainties (Romeo-Gamez et al., 2017). The most robust results can be found within CC, APt, EPm, ETt and POF where the ratio between the longest and shortest bandwidth lies below 200%. Apart from ETt, the probability distributions for those impact categories have a very similar shape (Figure 10.9). This can be attributed to the retrieval process which contributed most to those impact categories (Figure 10.2).

The highest uncertainties occur within ETfw, ETm and EPfw where the bandwidth variations exceed 1500%. Within those impact categories the highest uncertainty occurs in scenario 2 which may be linked to the electricity production.

Within WD, FD, MD and HT the ratio between longest and shortest bandwidth lies between 200% and 1500%. For WD and HT the highest uncertainties occur in scenario 2. Again, this is presumably linked to the overall uncertainties of the electricity production. The bandwidth of scenario 5 is much longer for FD (Figure 10.9) and can be attributed to the landfill process related diesel production. The higher uncertainty of scenario 1-3 in MD can be explained by the steel and lead recycling as this was missing in scenario 4 (Figure 10.2).

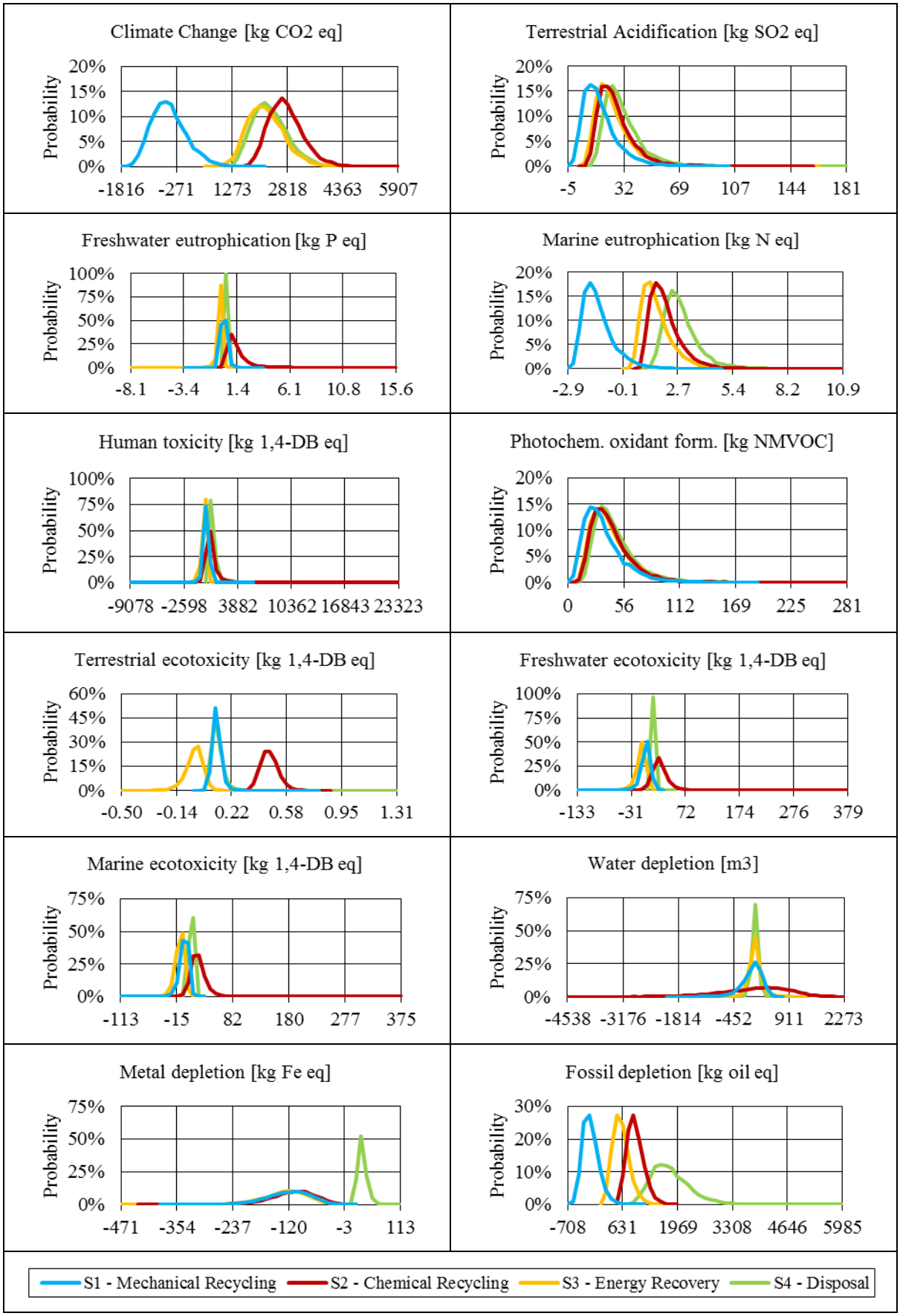


Figure 10.9: Probability distributions

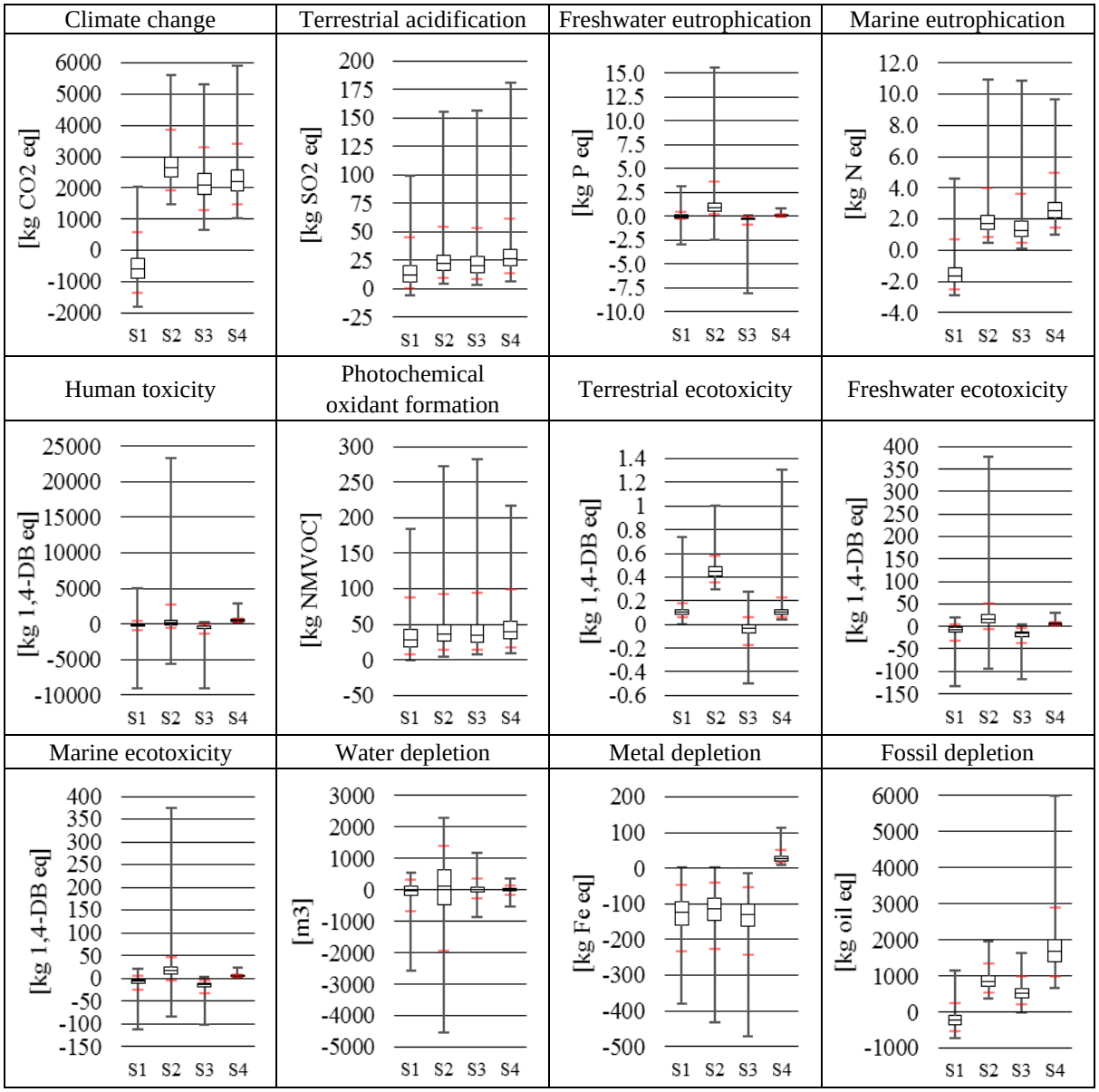


Figure 10.10: Uncertainty analysis results

10.5 Summary of main results

This chapter provided a detailed analysis of the potential environmental impacts for different DFG waste treatment scenarios. The impact contribution analysis highlighted that the mechanical recycling and energy recovery had a lower environmental impact compared to the chemical recycling and disposal scenario across all impact categories. While the uncertainty analysis revealed frequent overlaps of the scenarios' uncertainty ranges, it also provided a confidence level of 95% for the following results:

- CC: Mechanical recycling has lowest impact
- EP_{fw}: Energy recovery outperforms chemical recycling and disposal

- EP_m: Mechanical recycling outperforms chemical recycling and disposal
- HT, ET_m: Energy recovery outperforms disposal
- ET_t: Energy recovery has lowest and chemical recycling highest impact
- ET_{fw}: Energy recovery outperforms disposal
- MD: Disposal has highest impact
- FD: Mechanical recycling outperforms chemical recycling and disposal

The processes that contributed most to the environmental impacts within the scenarios are again summarised in Table 10.20. Especially the retrieval and gasification caused significant impacts across multiple impact categories. For the gasification this mainly derived from a high electricity consumption while the retrieval impacts were mainly caused by process emissions. Other important processes were the extrusion, incineration and lead recycling with significant credits from an avoided nylon, energy and lead production.

Table 10.20: Summary of processes with high environmental impact contributions

Contribution > 20%		Scenario	Unit	CC	AP_t	EP_{fw}	EP_m	HT	POF	ETt	ETfw	ETm	WD	MD	FD
Impacts	Retrieval	All	%	59	84		48		92						36
	Gasification	2	%	27		87		80		79	83	83			
	Incineration	3	%	26											
	Landfill	4	%				32	49							64
Credits	Extrusion	1	%	-89	-32		-97						-93		-52
	Incineration	3	%			-32		-30		-34	-30	-30			
	Steel recycling	1-3	%											-45	
	Lead recycling	1-3	%					-49			-44	-44		-50	

The sensitivity analysis investigated the effect of changes to the waste composition, energy production and transport distances as well as market and technology related substitution factors. While all input parameters changes caused fluctuations to the impact assessment results, only changes to the waste composition significantly altered the scenario ranking results.

More specifically, a waste composition comprising of trawl nets (no lead content) resulted in a significant human toxicity reduction of the disposal scenario while credits from the lead recycling in the other scenarios were also reduced. This allowed the disposal scenario to jump ahead of the chemical recycling scenario with a margin of more than 20%. As the reverse is also true, gillnets with a high lead content increased their human toxicity contribution significantly in the disposal scenario. The implications of this and other issues will be discussed in the next chapter.

11 Discussion

11.1 Areas for discussion

Having analysed the potential environmental impacts of DFG waste management scenarios in the previous chapters, this chapter discusses the research in a broader context. For this, the LCIA results are compared with other LCA studies from the literature, before its limitations and implications are further discussed.

11.2 Comparison with other LCAs

Scenario ranking results

The literature review in Chapter 3 revealed that LCA studies in the area of waste management generally support the waste hierarchy. However, the results of this thesis question this, because higher environmental impacts for the chemical recycling scenario were established in Chapter 10 compared to the energy recovery.

This discrepancy to the waste hierarchy can be explained by the comparatively high energy demand for the modelled chemical recycling scenario. Instead of 1.65 kWh/kg (Table 7.9), previous studies that supported the waste hierarchy adopted much lower energy demands such as 0.363 kWh/kg in Kreißig et al. (2003). The different energy demands can be explained by the process temperature of the underlying recycling techniques.

Studies with a low energy demand typically refer to low temperature pyrolysis techniques (i.e. Al-Salem, Evangelisti and Lettieri, 2014; Kreißig et al., 2003). However, at a commercial scale such processes only exist in combination with a much higher temperature incineration or gasification step (Chen et al., 2015). This means that studies in support of the waste hierarchy like Kreißig et al. (2003) may not be representative for large-scale applications. Although other studies like Shonfield (2008) or Al-Salem, Evangelisti and Lettieri (2014) have also modelled chemical recycling techniques with a similarly high energy demand as in this thesis, they did not directly compare them with an energy recovery. This means that further research is needed to clarify the position of energy intensive chemical recycling techniques in the waste hierarchy.

Process contributions

Previous studies showed that waste treatment processes cause relatively large environmental impacts while pre-treatment steps and the road transport play a less significant role (i.e. Faraca, Martinez-Sanchez and Astrup, 2019; Wäger and Hischier, 2015; Biganzoli et al., 2015). This was also confirmed in Chapter 10. Therefore, future LCAs may use those results to justify an exclusion of transport and pre-treatment steps from their system boundaries. However, as LCA studies are case-specific any process exclusion should be treated with extreme care.

Sensitivity of input parameters

The influence of input parameters on LCA results are frequently explored. For example, Faraca, Martinez-Sanchez and Astrup (2019) identify technology and market assumptions as the most influential parameters, while Stamford and Azapagic (2014; 2018) reveal that changes to the energy production mix can have large effects as well. In this thesis, those parameters were also altered as part of the sensitivity analysis (Chapter 10), but they did not change the scenario rankings at a significant scale. Thus, further research is needed to better understand under which conditions a relevant change to the scenario ranking occurs.

Apart from the energy, technology and market assumptions other sensitivity cases were evaluated as well. While the expected low sensitivity of the transport process (Laurent et al., 2014a) was confirmed, changes to the waste composition caused the largest fluctuations to the results. This is interesting, because the importance of the waste composition is debated in the literature. While Quirós et al. (2015) underline its high sensitivity especially for the incineration process, Wäger and Hischier (2015) report no significant effect on the overall results. Again, this shows that LCA studies are case specific and that decisions to disregard certain parameters from the analysis should be taken with care.

11.3 Research limitations

11.3.1 Modelling limitations

Modelling framework

For this thesis an attributional LCA was used. However, as the results may guide stakeholders to decide which waste infrastructure to set up for DFG in the future, a consequential LCA may have also been used. This was not done, because the setup of a small-scale DFG processing

plant was not expected to alter the current secondary materials or energy production at a significant scale. In other words, the difference between a consequential and attributional modelling would have been very small.

System boundaries

Capital goods and the landfilling of residues were excluded from the system boundaries due to their expected low environmental impact within the waste treatment scenarios (Chapter 4.2.3). However, as the exclusion favours the recycling and energy recovery, those aspects should be included in the future so that a fairer comparison can be made.

Within the system boundaries of this thesis four DFG waste management scenarios were compared. However, it would help to relate those findings to a prevention or default scenario during which no retrieval and waste management needs to take place. To allow this, new impact categories such as ghost fishing or the release of microplastics to the environment could be developed in the future. While some effort is already undertaken in this area (Sonnemann and Valdivia, 2017; Woods, Rødder and Verones, 2019) it will still take some time for those new impact categories to become commonly accepted and applied in LCA.

Long-term impacts

Due to high uncertainties, long-term emissions that continue to be emitted after 100 years – for example CO_2 emissions from a polymer degradation in landfills – were excluded from the impact assessment. Although this is commonly recommended (European Commission, 2010), it significantly improves the environmental profile of landfills. In fact, only 1% of the carbon content and 0.133% of the lead content were modelled to be released within this period (Chapter 9.2). Thus, considering a longer time frame would further increase the environmental impacts of the disposal scenario.

Non-environmental factors

Apart from environmental impacts, social and economic issues also need to be considered for a sustainability focused decision making (Stamford and Azapagic, 2014). For example, as diving operations and the handling of hazardous materials like toxic lead impose severe health and safety risks, potential worker injuries may be investigated more closely in the future. In addition, further attention should be paid on the economic sustainability of DFG waste management. In fact, without additional subsidies, the economic sustainability of the scenarios is in doubt,

because the presumed lack of ownership and thus a missing gate fee may be problematic even if a premium price for marine litter products could be charged.

11.3.2 Experimental limitations

Data collection

The measured in- and output weights from the experiments did not always add up. While this was attributed to losses in form of water evaporation, material dropping on the floor or residues in machines that could not be removed, it led to several assumptions and estimates. To get more robust data, water content measurements should be taken more regularly in future experiments.

Sample representativeness

Throughout the thesis the content of heavy metals, chlorine and other elements in DFG samples were determined by external partners. This data was used to establish the waste composition and to model process emissions for example from landfilling. However, for the analysis not always representative samples may have been used. For example, the rough shredded gillnet material revealed that lead accumulated at the bottom of the big bag – an area from which samples had not been taken. While this was accounted for in this thesis, the material should be better mixed before samples are taken.

Process variations

The modelled recycling scenarios mirrored the type and sequence of processes from the conducted experiments. However, the literature review revealed that other processes combinations are also possible. For example, instead of using centrifuges, an air blowing technique could be employed to separate lead from polymers (Jung et al., 2010). Also, for the chemical recycling a pyrolysis or depolymerisation process (Aquafil, n.d.) could be used. While this was out of scope for this study, such process variations could be evaluated in the future to get an even broader view.

11.3.3 Data limitations

Data availability

The limited availability of data is a common problem for LCAs, especially at an early stage for new processes or novel waste streams (Hetherington, 2014). While much primary data could be collected through experiments in this thesis, gaps in the LCI needed to be filled with secondary data. However, apart from being less representative, secondary data also varied in their

completeness for emission data. For example, carbon monoxide emissions were made available for the retrieval, steel recycling, transport and incineration process, but not for the lead recycling or landfill (Appendix B). While the available data was included in this thesis, data gaps should be filled as soon as new data is released.

Data representativeness

Within the modelled processes the data representativeness varies. For example, while most LCI models were built on data from 2017 to 2019, the landfill (DEFRA, 2004) and incineration (BREF, 2006) referred to older publications and thus waste treatment sites. Also, the gasification process reassembled a pilot plant whereas other processes like the extrusion, incineration and landfilling were based on commercial settings. In addition, and unlike processes that were fully based on DFG compositions, the emissions of the incineration and landfill process were partly modelled based on municipal solid waste. As this is again linked to data availability, the LCIs should be updated as soon as more representative data becomes available.

Waste composition

The DFG waste composition was modelled based on experimental data and as a range between trawl and gillnets (Appendix A). However, Chapter 2.2.3 revealed that DFG compositions can vary much more. For example, the lead content in DFG from the Northwest Straits Foundation (2015) was more than four times higher than modelled here. This is problematic, because already small changes to the waste composition had significant effects on the results (Chapter 10.3). While the modelled waste composition represents the findings from the area under study, effects of higher variations in the waste composition should be investigated more.

Electricity mix

Like for the waste composition, the scope of the electricity mix could be expanded more. This is because the modelled electricity mix is based on Germany which differs substantially from other European countries like France or Norway. While a sensitivity analysis was conducted to model a future and more renewable energy mix, it did not encompass the large variety of different energy mixes in Europe. It would be beneficial to model other countries with a higher share of for example hydro, solar or nuclear power to see whether the chemical recycling scenario could outperform the energy recovery then.

11.4 Research implications

Implications for the LCA community and wider research field

The literature review identified a gap between marine litter collections and waste treatment techniques. Using LCA and systems thinking proved very suitable to bridge the gap between the two domains. Although a DFG waste management system is not yet established in Europe, the use of LCA in this thesis may support the planning from an environmental point of view. Especially the use of sensitivity and uncertainty analysis can help to focus on critical issues early on. While in this thesis the waste composition could be identified as the most sensitive parameter, the transport distance did not play an important role. To confidently reach similar conclusions in other LCAs, the use of sensitivity and uncertainty analysis are further encouraged.

Implications for fishers, authorities and port-reception facilities

As this thesis showed, DFG can continue to catch fish making it a direct competitor for existing fishers. While additional education and awareness campaigns may help to share this knowledge, fishers usually understand the severity of DFG. Still, lost fishing gear is seldomly reported as fishers are presumably afraid of covering the clean-up costs. Norway solves this dilemma by sharing part of the recovery cost and by organising an annual retrieval campaign that returns undamaged fishing gear back to their owners. This builds up trust and encourages the fishers to share the positions of the lost fishing gear. While taxpayers are indirectly charged for this service, it allows for a much more efficient retrieval process to take place. As otherwise individual retrieval efforts have a significant environmental impact, the introduction of a regular retrieval campaign like in Norway may be considered for the rest of Europe as well.

Waste from ships need to be adequately managed in ports (Directive (EU) 2019/883, 2019). However, as DFG is considered historical and not ship-based waste by most fishers and harbour personnel, harbours are usually not equipped to store or handle DFG (Press, 2017). Still, to avoid possible toxic lead to be mixed with commercial or household waste, a separate collection in harbours should take place. In fact, to allow for valuable items such as lead lines to be directly reused in ports, not only a storage but also a pre-treatment should take place. A removal of fish, heavy and bulky items would also facilitate the following transport and waste treatment steps. Ideally fishers could help to carry out this task as their experience in handling fishing gear gives them an advantage over other staff. Involving fishers may also increase awareness while providing an additional income stream.

Implications for waste management companies

The expected annual quantity of DFG is very small so that the setup of a new waste treatment plant for DFG may not be justified. Still, this thesis established the mechanical recycling and energy recovery as the least environmentally harmful waste treatment options. As such infrastructure already exists for EOLFG and because EOLFG and DFG have a similar waste composition, it may be possible to integrate DFG. However, for this a pre-treatment step would be required to reduce the size and amount of contamination in DFG. Depending on the material quality DFG could be separated into a mechanical recycling and energy recovery suitable fraction. If this is not possible in harbours, another agent would be needed to carry out this task. To ensure that pre-treated DFG is appropriate for a processing, clear waste acceptance criteria would need to be established as well.

The least preferable waste treatment options were the chemical recycling and disposal scenario. Especially for lead rich gillnets a disposal should be avoided as this can otherwise cause significant human health impacts. The chemical recycling was currently too energy intensive to be environmentally competitive. However, as process improvements may change this in the future, such technologies should not be neglected but continuously monitored.

Implications for policy makers

Gasification or other chemical recycling techniques like pyrolysis have the potential to turn plastic waste into new products. However, as the products can also be used to generate heat or electricity, there is a debate whether to classify chemical recycling as an energy recovery or recycling technique. In context of the increasing legal recycling targets, this is important for companies to decide whether to purchase such machines. As this thesis questioned the gasification's ability to compete with incineration plants from an environmental perspective, policy makers should continue to request detailed LCAs before counting novel technologies as recycling techniques.

This thesis highlighted that the retrieval and waste treatment of DFG is very resourceful and technical challenging. While this means that additional funding or guidance may be required for waste management companies to adequately process DFG, it also indicates that more focus should be paid on preventive measures to stop fishing gear from becoming derelict. As a first step the already proposed indirect waste fee in port reception facilities and the extended producer responsibility [EPR] for fishing gear (Chapter 2.1.4) should be introduced.

Particularly the EPR scheme would be beneficial as it clearly imposes the fishing gear collection and waste treatment cost onto the fishing gear producer. This would encourage them to setup an effective end-of-life fishing gear collection system, possibly through a deposit return scheme, to make sure that the collection cost is reduced. It would also motivate them to come up with a better fishing gear design comprising of fewer and less harmful materials to facilitate repair, reuse and recycling approaches.

12 Conclusion

12.1 Rationale

Derelict fishing gear is one of the most abundant and harmful types of marine litter that is increasingly retrieved from the ocean. However, after landing there is currently no established waste treatment system for this type of waste in Europe. As part of this thesis and in collaboration with WWF Germany, PreZero and other partners, work was conducted to evaluate alternatives to the default disposal of DFG in landfills.

12.2 Contribution to knowledge

In line with the overall research aim - to identify the most environmentally sustainable waste treatment option for DFG -, this thesis established and compared the potential environmental impacts of alternative waste management pathways for DFG. As part of this, three research objectives were addressed.

The first research objective - to illustrate the current knowledge on DFG waste management - was addressed by providing a systematic literature review on DFG retrievals and their waste treatment as well as on relevant LCA studies in the field. This information is particularly relevant for stakeholders that plan similar clean-up projects.

The second research objective - to investigate the technical feasibility of DFG management options - was addressed by carrying out and critically describing industrial recycling experiments for DFG. This involved several pre-treatment, extrusion and gasification trials. Primary data was collected to compile a process flow chart and an indicative DFG waste composition. This information is not only important for a quantitative comparison of the options, but also for waste companies to evaluate whether they can accept this type of waste.

The third research objective - to establish the potential environmental impacts for DFG management options - was addressed by conducting a Life Cycle Assessment for the (1) mechanical recycling, (2) chemical recycling, (3) energy recovery and (4) disposal of DFG. As part of this novel inventory data was compiled and a Life Cycle Impact Assessment was conducted showing that the mechanical recycling and energy recovery outperformed the chemical recycling and disposal scenario. Those findings are particularly relevant for guiding future actions in this field.

12.3 Recommendations

Throughout this thesis, several recommendations for improvements were made. Some of the most important points are summarised again.

Prevention

To avoid or improve DFG's severe impacts inside the ocean as well as its complicated retrieval and waste management, preventive measures should be prioritised. Those may include to:

- Implement an extended producer responsibility scheme for fishing gear
- Improve fishing gear design (i.e. replace lead, reduce polymer types)
- Increase awareness by involving fishers

Retrieval

To reduce the environmental impact of the retrieval it is recommended to:

- Encourage the reporting of lost fishing gear
- Combine multiple suspected DFG locations in an annual retrieval campaign

Pre-treatment

To manage DFG effectively and to stop toxic lead from getting mixed with other waste and thus reaching incineration plants or landfill sites, the following steps are proposed:

- Collect DFG in separate containers
- Carry out sorting and possibly other steps to reduce contaminants in DFG
- Involve fishers to speed-up the process and to identify valuable components for a reuse
- Distinguish material grades for further mechanical recycling or energy recovery

Waste treatment

When establishing waste treatment processes for DFG, the following should be considered:

- Avoid the disposal of DFG that contains lead
- Adapt EOLFG mechanical recycling and energy recovery facilities to accept DFG
- Establish waste acceptance criteria and communicate them with pre-treatment steps
- Monitor chemical recycling technologies for their future suitability

12.4 Future work

As briefly discussed in Chapter 11, this thesis opens many routes for further pursuit. Some of the most important aspects are summarised here.

Incremental future work

Future work could focus on improving the overall data quality. To increase the accuracy of the results, data from outdated sources should be replaced with newer information. Particularly for the incineration, an updated version of the BREF is now available. Furthermore, missing emission data for relevant substances should be added to the process inventories as soon as they become available. It is also desirable to expand the system boundaries to include capital goods and secondary landfills to strengthen the results.

Future work could also explore the effect of a changing waste composition to broaden the current focus on DFG from the Baltic Sea. As particular sensitive parameter, the waste composition could then be used to indicate, when changes to the scenario ranking occur. In other words, for each waste treatment scenario it may be possible to identify a specific waste mix that is most preferable possibly guiding decision making in this field.

Similarly, to the waste composition, it would be desirable to further evaluate the conditions under which a chemical recycling becomes more preferably than an energy recovery. For this, the electricity consumption of the investigated gasification process may be reduced in a sensitivity analysis or the country for the DFG treatment and thus the underlying energy mix may be changed.

Future work could also address non-environmental aspects. Especially the economic sustainability of the process should be further investigated to determine whether additional subsidies are needed to process DFG. In fact, the system boundaries are already established, so that a life cycle costing analysis would not require too much additional work. To pursue this, further information about machine and operating costs would be needed.

Radical future work

Long-term orientated future work could be carried out to reduce the impact of DFG in the ocean. For example, fishing gear could be equipped with sensors to locate and retrieve them faster after they have been lost, reducing the exposure time and risk of entanglements to marine life. A shorter exposure time could also be achieved through biodegradable fishing gear. However,

given the high product requirements for fishing gear, it is unclear whether a suitable replacement for virgin polymers could be developed. Another way of reducing the negative impacts of DFG in the ocean would be to identify innovative fishing techniques that reduce the potential loss of fishing gear. For example, instead of a bottom trawling which can lead to gear losses at underwater obstacles, a fishing technique based on light to attract and guide fish into a net could be used.

Other long-term orientated work could focus on improving the recyclability of complex waste mixes like DFG. For example, sensor-based sorting machines could be developed to better separate mixed and entangled polymers. If possible, such future technologies could also be placed onboard of special ships to immediately treat this waste after the retrieval.

12.5 Closing words

Based on experiments and reference data, this thesis established the environmental profile of different waste treatment options for derelict fishing gear. While the results indicate that a mechanical recycling and energy recovery are the most advantageous, a large-scale implementation is still missing to validate those results.

DFG proved to be a particularly contaminated and diverse mix of materials that is labour-intensive and time-consuming to process. While an effort should be made to treat DFG in the most environmentally friendly way, the key is to intensify preventive measures to stop the accumulation of fishing gear or other waste in the sea.

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Appendix A: Calculation of the waste composition

Table A1 provides a summary of the relative in- and output data from Chapter 5-7. For example, the retrieval of gillnets resulted in an output of 3.4% fish and 0.5% material loss while the sorting of gillnets removed 6.2% steel, 2% minerals and 0.5% material loss (Table A1).

Table A.1: Summary of the experimental in- and output flows

Process	Unit	Gillnets								Trawl nets							
		Input	Output							Input	Output						
			Organics	Steel	Lead	Minerals	PE / PP	PA	Loss		Organics	Steel	Lead	Minerals	PE / PP	PA	Loss
Retrieval	%	100	3.4						0.5	100	0.0						0.5
Sorting	%	100		6.2		2.0			0.5	100		6.2		11.9			7.7
Rough Shredding	%	100		1.5					-0.4	100		1.5					8.7
Fine Shredding	%	100		0.0					4.1	100		0.0					7.6
1. Density Separation	%	100			2.6	16.6			1.0	100			0.0	37.5			1.0
2. Density Separation	%	100					7.8		1.0	100					0.0		1.0
Washing	%	100				8.2		50.0	24.8	100				23.3		15.0	-17.4
Remaining	%	100				1.0		99.0		100				0.6		99.4	
Gasification	%	100			32.4	32.4	3.9	30.4	1.0	100							

To determine the overall waste composition the process specific output data is converted based on their relative input contribution. For example, the sorting of gillnets represents 96.1% of the DFG input because 3.9% of other materials were already removed (Table A1). Therefore, the overall steel and minerals output from the sorting is 1.9% and 6.0% while the relative material loss remained at 0.5% (Table A2). The contributions are then added up for each material group to give the total waste composition (Table A2).

Table A.2: Converted in- and output flows in relation to the retrieval input in [%]

Process	Gillnets									Trawl nets							
	Split	Input	Output							Input	Output						
			Organics	Steel	Lead	Minerals	PE / PP	PA	Loss		Organics	Metal	Lead	Minerals	PE / PP	PA	Loss
Retrieval		100	3.4						0.5	100	0.0						0.5
Sorting		96.1		6.0		1.9			0.5	99.5		6.2		11.9			7.7
Rough Shredding		87.7		1.3					-0.3	73.7		1.1					6.4
Fine Shredding		86.7							3.6	66.2		0.0					5.0
1. Density Separation	54.8	45.6			1.2	7.6			0.5	61.1			0.0	23.0			0.6
2. Density Separation		36.4				0.0	2.8		0.4	37.6					0.0		0.4
Washing		33.2				2.7		16.6	8.2	37.2				8.7		5.6	-6.5
Remaining		5.65				0.1		5.6		29.4				0.2		29.2	
Gasification	45.2	37.6			12.1	12.1	1.5	11.4	0.4								
Total			3.4	7.3	13.3	24.4	4.3	33.6	13.6		0	7.3	0	43.7	0	34.8	14.2

After shredding the gillnet fractions were divided into 292 kg for the density separation (54.8%, assuming a water content of 3%) and into 311 kg for the gasification (45.2%, assuming a water content of 25%). As this affects the relative input, the split was considered when calculating the overall waste composition (Table A2).

Previously made assumptions (Chapters 5-7) are used to allocate the material loss to the remaining material groups. Those assumptions include that all material loss from the retrieval and sorting are allocated to the residue fraction (Chapter 5) and that the remaining material loss is allocated to represent the share of materials still available in the process input. For example, the allocation of the material loss for the rough shredding of trawl nets is calculated like this:

- Steel $= \frac{Steel\ left}{Known\ Input\ left} = \frac{7.3-6.2}{(7.3-6.2)+(43.7-11.9)+(34.8-0)} \sim 1.7\%$
- Minerals $= \frac{Minerals\ left}{Known\ Input\ left} = \frac{43.7-11.9}{(7.3-6.2)+(43.7-11.9)+(34.8-0)} \sim 46.9\%$
- Nylon $= \frac{Nylon\ left}{Known\ Input\ left} = \frac{34.8-0}{(7.3-6.2)+(43.7-11.9)+(34.8-0)} \sim 51.4\ \%$

The same principle is used for the allocation of the remaining material loss. The calculation for gillnets is based on the mechanical recycling pathway as this provided the most representative and detailed waste composition. The results of all allocation keys are provided in Table A.3.

Table A.3: Loss allocation

Process	Unit	Gillnets							Trawl nets						
		Loss	Allocation						Loss	Allocation					
			Organics	Steel	Lead	Minerals	PE / PP	PA		Organics	Steel	Lead	Minerals	PE / PP	PA
Retrieval	%	0.5	-			100			0.5				100		
Sorting	%	0.5				100			7.7				100		
Rough Shredding	%	-0.3		3.5	3.1	27.4	7.5	58.5	6.4		1.7		46.9		51.4
Fine Shredding	%	3.6			3.2	28.4	7.7	60.7	5.0				47.7		52.3
1. Density Separation	%	0.5			3.2	28.4	7.7	60.7	0.6				47.7		52.3
2. Density Separation	%	0.4				10.0	10.2	79.8	0.4				20.2		79.8
Washing	%	8.2				11.1		88.9	-6.5				20.2		79.8
Gasification	%	0.4			3.2	28.4	7.7	60.7							

The total waste composition without a material loss is determined in Table A4. This calculation resulted in an average waste composition of 1.7% organics, 7.4% steel, 6.8% lead, 41.9% minerals and 42.2% polymers providing the basis for the functional unit (Chapter 4.2.2).

Table A.4: Waste composition without material loss

Material contribution	Gillnets						Trawl nets					
	Organics	Steel	Lead	Minerals	PE / PP	PA	Organics	Steel	Lead	Minerals	PE / PP	PA
Known materials	3.4	7.3	13.4	24.5	4.3	33.6	0	7.3	0	43.7	0	34.8
Material loss	13.6						14.2					
Retrieval	-			0.5						0.5		
Sorting				0.5						7.7		
Rough Shredding		<<	<<	-0.1	<<	-0.2		0.1		3.0		3.3
Fine Shredding			0.1	1.0	0.3	2.2				2.4		2.6
1. Density Separation			<<	0.1	<<	0.3				0.3		0.3
2. Density Separation				<<	<<	0.3				0.1		0.3
Washing				0.9		7.3				-1.3		-5.2
Gasification			<<	0.1	<<	0.2						
Total	3.4	7.3	13.5	27.5	4.6	43.6	0.0	7.4	0.0	56.3	0.0	36.2

Appendix B: Data selection from the Ecoinvent database

To model the ancillary materials, energy and avoided production from the unit processes, the following Ecoinvent processes were selected (Table B.1).

Table B.1: Ecoinvent processes for ancillary materials and avoided production

Process	Flow	Selected Ecoinvent process
1	4-methyl-2-pentanone	4-methyl-2-pentanone {GLO}
5	PolySepar CFX 1088	Aluminium hydroxide {GLO}
11	Flocculation agents	Aluminium hydroxide {GLO}
11	Ammonia	Ammonia, liquid {RER}
9	Argon	Argon, liquid {GLO}
5	PolySepar PK 1455	Cationic resin {GLO}
11	Precipitating agents	Cationic resin {GLO}
9,10	Charcoal	Charcoal {GLO}
11	Activated carbon	Charcoal {GLO}
1	Cobalt	Cobalt {GLO}
1	Copper oxide	Copper oxide {GLO}
1,12,13	Diesel	Diesel {Europe without Switzerland}
2-12	Electricity (water)	electricity production, hydro, run-of-river
	Electricity (biomass)	heat and power co-generation, biogas, gas engine
	Electricity (wind)	electricity production, wind, >3MW turbine, onshore
	Electricity (solar)	photovoltaic, 570kWp open ground installation, multi-Si
	Electricity (natural gas)	electricity production, natural gas, combined cycle power plant
	Electricity (lignite)	electricity production, lignite
	Electricity (hard coal)	electricity production, hard coal
	Electricity (nuclear)	electricity production, nuclear, pressure water reactor
1	Ethanol	Ethanol, without water, in 99.7% solution state, from ethylene {GLO}
1	Ethyl benzene	Ethyl benzene {GLO}
9	Graphite electrodes	Graphite {GLO}
6	Heat	Heat, in chemical industry {RER}
8,11	HCL	Hydrochloric acid, without water, in 30% solution state {RER}
10	Lead	Lead {GLO}
8,9,11	Lime	Lime {GLO}
9	Low-alloyed steel	Steel, unalloyed {GLO}
1,13	Lubricant oil	Lubricating oil {GLO}
9,10	Natural gas	Natural gas, high pressure {DE}
8,9	Nitrogen	Nitrogen, liquid {RER}
7	Nylon	Nylon 6-6 {GLO}
9,10	Oxygen	Oxygen, liquid {RER}
9	Pig / Direct reduced iron	Pig iron {GLO}
9,10	Refractory lining	Refractory, basic, packed {GLO}
10	Sodium bicarbonate	Soda ash, dense {GLO}
4	Salt	Sodium chloride, powder {GLO}
10,11	Sodium hydroxide	Sodium hydroxide, without water, in 50% solution state {GLO}
9	Alloys	Steel, unalloyed {GLO}
8	Syngas	Synthetic gas {CH}
4,5,8-11	Water	Tap water {Europe without Switzerland}
1	White spirit	White spirit {GLO}
1	Xylene	Xylene {GLO}
1	Zinc oxide	Zinc oxide {GLO}

1- Retrieval, 2- Sorting, 3- Shredding, 4- Density Separation, 5- Washing, 6- Drying, 7- Extrusion, 8- Gasification, 9- Steel recycling, 10- Lead recycling, 11- Incineration, 12- Landfill, 13- Transport

An overview of the modelled emission data is provided in Table B.2.

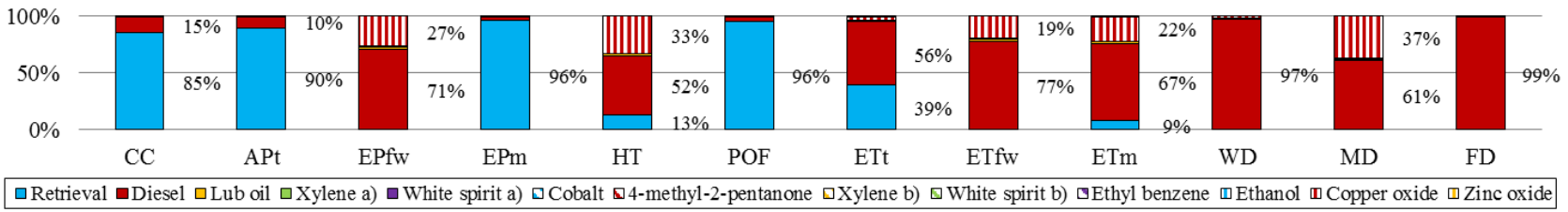
Table B.2: Modelled process emissions to air and water

Emissions to air	Relevant Process						Emissions to water	Relevant Process			
	1	9	10	11	12	13		1	10	11	12
1,1-dichloroethane					x		4-methyl-2-pentanone	x			
Ammonia				x		x	AOX		x		
Arsenic	x		x	x	x	x	Antimony compounds		x		
Benzene		x			x		Arsenic		x	x	x
Benzo(a)pyrene				x		x	Cadmium compounds		x	x	
Benzo(b)fluoranthene				x		x	Chlorides			x	x
Benzo(k)fluoranthene				x		x	Chromium			x	x
Cadmium	x	x	x	x	x	x	Cobalt compounds	x			
Carbon dioxide [a)]	x	x	x	x	x	x	COD		x	x	
Carbon monoxide [a)]	x	x		x		x	Copper compounds	x	x	x	x
Chlorobenzene		x			x		Ethanol	x			
Chloroethane					x		Ethyl benzene	-			
Chloroethene					x		Fluoride				x
Chromium	x	x		x		x	Hydrocarbons		x		
Copper	x	x		x		x	Iron		x		
Hexachloro-Benzene	x			x			Lead compounds		x	x	x
Hydrogen chloride		x		x	x		Mercury compounds		x	x	
Hydrogen fluoride		x		x	x		Nickel compounds		x	x	x
Indeno(1,2,3-cd)pyrene				x		x	Nitrogen		x	x	x
Lead	x	x	x	x	x	x	Organo-tin				x
Mercury	x	x		x	x	x	Phosphorous compounds		x		x
Methane					x	x	Silver		x		
Nickel	x	x		x	x	x	Sulfallate			x	
Nitrogen oxides	x	x		x	x	x	Suspended solids		x		
Nitrous oxide [b)]						x	Thallium		x		
NMVOC	x			x	x	x	Tin		x		
PAH		x					TOC				x
Particulates	x	x	x	x	x	x	White spirit	-			
PCB	x	x	x	x			Xylene	x			
PCDD/F [c)]	x	x	x	x	x		Zinc compounds	x	x	x	x
Selenium	x			x		x	Comments:				
Sulphur dioxide	x	x		x		x	[a)]: Modelled from fossil sources.				
Sulphur oxides			x		x		[b)]: Modelled as Dinitrogen monoxide.				
Tetratchloroethene					x		[c)]: Modelled as unspecified Dioxins.				
TOC											
Zinc	x	x	x	x		x					

1- Retrieval, 9- Steel recycling, 10- Lead recycling, 11- Incineration, 12- Landfill, 13- Transport

Table C.1: Characterisation results for scenario 1

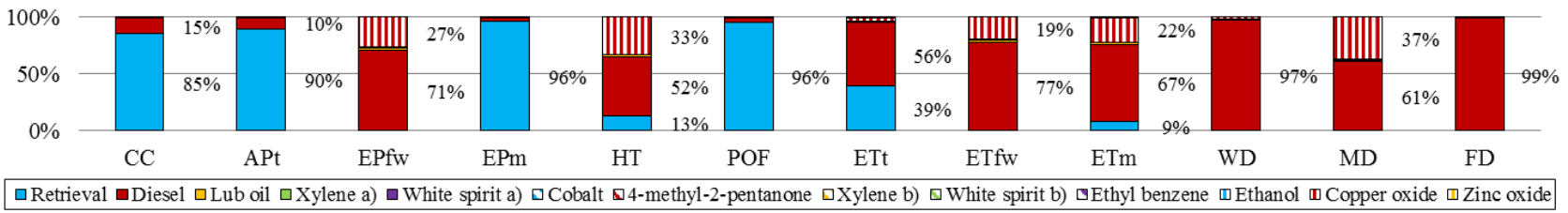
Impact category		CC	APt	EPfw	EPm	HT	POF	ETt	ETfw	ETm	WD	MD	FD
Unit		kg CO_2 eq	kg SO_2 eq	kg P eq	kg N eq	kg 14-DB eq	kg NMVOC	kg 14-DB eq			m3	kg Fe eq	kg oil eq
1)	Retrieval	**9.26E+01**	**1.34**	0	**8.93E-02**	**6.39E-01**	**2.39 E+00**	**1.28E-03**	9.69E-04	**1.10E-02**	0	0	0
2)	Diesel	**1.58E+01**	**1.45E-01**	**1.96E-03**	3.24E-03	**2.57**	9.90E-02	**1.84E-03**	**1.05E-01**	**8.52E-02**	**1.89E-01**	**4.97E-01**	**3.74E+01**
	Lubricating oil	1.79E-01	1.38E-03	5.23E-05	3.81E-05	6.16E-02	2.54E-03	1.93E-05	2.64E-03	2.35E-03	1.89E-03	1.26E-02	2.96E-01
	Xylene [a)]	4.94E-03	1.31E-05	6.43E-08	3.34E-07	8.80E-05	1.53E-05	2.28E-07	3.86E-06	6.38E-06	8.25E-05	1.20E-05	4.21E-03
	White spirit [a)]	2.98E-02	2.42E-04	1.00E-05	6.89E-06	1.14E-02	4.53E-04	3.35E-06	4.88E-04	4.34E-04	3.46E-04	2.38E-03	5.29E-02
	Cobalt [a)]	1.99E-04	2.00E-06	7.77E-08	2.04E-07	9.14E-05	2.04E-06	1.58E-08	2.69E-06	2.55E-06	3.31E-06	4.12E-05	4.88E-05
	4-methyl-2 -pentanone [a)]	2.35E-03	1.04E-05	3.97E-07	2.73E-07	2.88E-04	2.07E-05	7.76E-08	1.33E-05	1.03E-05	4.23E-05	3.88E-05	1.29E-03
	Xylene [b)]	7.82E-03	2.08E-05	1.02E-07	5.30E-07	1.39E-04	2.42E-05	3.61E-07	6.11E-06	1.01E-05	1.31E-04	1.90E-05	6.66E-03
	White spirit [b)]	6.59E-05	5.37E-07	2.22E-08	1.53E-08	2.52E-05	1.00E-06	7.41E-09	1.08E-06	9.62E-07	7.67E-07	5.28E-06	1.17E-04
	Ethyl benzene [b)]	3.11E-03	1.15E-05	9.20E-07	4.33E-07	7.47E-04	1.62E-05	1.04E-07	2.67E-05	2.52E-05	4.92E-05	8.36E-05	2.16E-03
	Ethanol [b)]	6.70E-04	2.23E-06	2.72E-07	6.74E-08	9.59E-05	3.44E-06	2.14E-08	4.10E-06	3.83E-06	6.11E-06	2.14E-05	5.47E-04
	Copper oxide [b)]	7.48E-02	3.10E-03	**7.31E-04**	1.87E-04	**1.63 E+00**	7.78E-04	1.22E-04	**2.62E-02**	**2.79E-02**	2.28E-03	**3.02E-01**	2.21E-02
	Zinc oxide [b)]	1.08E-02	3.41E-05	2.17E-06	1.14E-06	4.21E-03	2.30E-05	1.05E-05	1.64E-04	2.02E-04	4.04E-05	6.35E-04	3.73E-03



[1)] Emissions occur during the process. [2)] Emissions occur during the material production. [a)] Derives from boat paint. [b)] Derives from antifouling.

Table C.2: Characterisation results for the Retrieval per hour

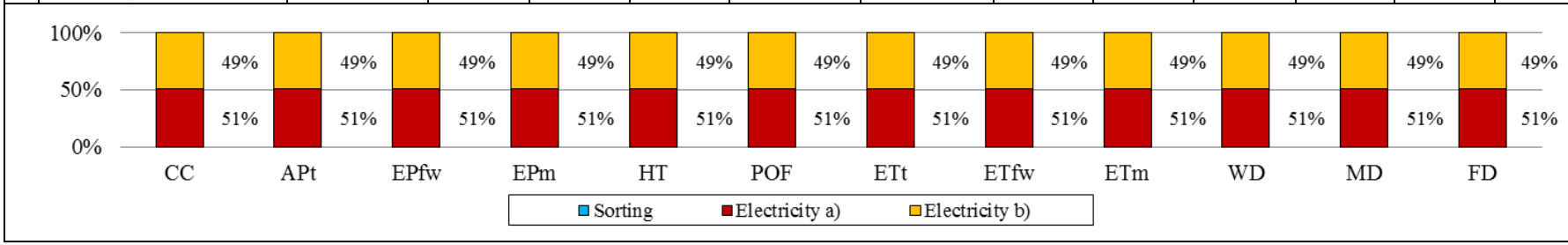
Impact category		CC	APt	EPfw	EPm	HT	POF	ETt	ETfw	ETm	WD	MD	FD
Unit		kg CO_2 eq	kg SO_2 eq	kg P eq	kg N eq	kg 14-DB eq	kg NMVOC	kg 14-DB eq			m3	kg Fe eq	kg oil eq
1)	Retrieval	**9.26E+01**	**1.34**	0	**8.93E-02**	**6.39E-01**	**2.39 E+00**	**1.28E-03**	9.69E-04	**1.10E-02**	0	0	0
2)	Diesel	**1.58E+01**	**1.45E-01**	**1.96E-03**	3.24E-03	**2.57**	9.90E-02	**1.84E-03**	**1.05E-01**	**8.52E-02**	**1.89E-01**	**4.97E-01**	**3.74E+01**
	Lubricating oil	1.79E-01	1.38E-03	5.23E-05	3.81E-05	6.16E-02	2.54E-03	1.93E-05	2.64E-03	2.35E-03	1.89E-03	1.26E-02	2.96E-01
	Xylene [a)]	4.94E-03	1.31E-05	6.43E-08	3.34E-07	8.80E-05	1.53E-05	2.28E-07	3.86E-06	6.38E-06	8.25E-05	1.20E-05	4.21E-03
	White spirit [a)]	2.98E-02	2.42E-04	1.00E-05	6.89E-06	1.14E-02	4.53E-04	3.35E-06	4.88E-04	4.34E-04	3.46E-04	2.38E-03	5.29E-02
	Cobalt [a)]	1.99E-04	2.00E-06	7.77E-08	2.04E-07	9.14E-05	2.04E-06	1.58E-08	2.69E-06	2.55E-06	3.31E-06	4.12E-05	4.88E-05
	4-methyl-2 -pentanone [a)]	2.35E-03	1.04E-05	3.97E-07	2.73E-07	2.88E-04	2.07E-05	7.76E-08	1.33E-05	1.03E-05	4.23E-05	3.88E-05	1.29E-03
	Xylene [b)]	7.82E-03	2.08E-05	1.02E-07	5.30E-07	1.39E-04	2.42E-05	3.61E-07	6.11E-06	1.01E-05	1.31E-04	1.90E-05	6.66E-03
	White spirit [b)]	6.59E-05	5.37E-07	2.22E-08	1.53E-08	2.52E-05	1.00E-06	7.41E-09	1.08E-06	9.62E-07	7.67E-07	5.28E-06	1.17E-04
	Ethyl benzene [b)]	3.11E-03	1.15E-05	9.20E-07	4.33E-07	7.47E-04	1.62E-05	1.04E-07	2.67E-05	2.52E-05	4.92E-05	8.36E-05	2.16E-03
	Ethanol [b)]	6.70E-04	2.23E-06	2.72E-07	6.74E-08	9.59E-05	3.44E-06	2.14E-08	4.10E-06	3.83E-06	6.11E-06	2.14E-05	5.47E-04
	Copper oxide [b)]	7.48E-02	3.10E-03	**7.31E-04**	1.87E-04	**1.63 E+00**	7.78E-04	1.22E-04	**2.62E-02**	**2.79E-02**	2.28E-03	**3.02E-01**	2.21E-02
	Zinc oxide [b)]	1.08E-02	3.41E-05	2.17E-06	1.14E-06	4.21E-03	2.30E-05	1.05E-05	1.64E-04	2.02E-04	4.04E-05	6.35E-04	3.73E-03



[1)] Emissions occur during the process. [2)] Emissions occur during the material production. [a)] Derives from boat paint. [b)] Derives from antifouling.

Table C.3: Characterisation results for the Sorting per kg DFG_1

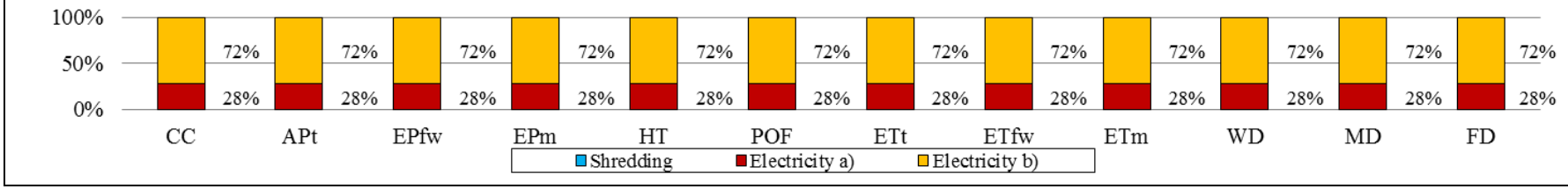
	Impact category	CC	APt	EPfw	EPm	HT	POF	ETt	ETfw	ETm	WD	MD	FD
	Unit	kg CO_2 eq	kg SO_2 eq	kg P eq	kg N eq	kg 14-DB eq	kg NMVOC	kg 14-DB eq			m3	kg Fe eq	kg oil eq
1)	Sorting	0	0	0	0	0	0	0	0	0	0	0	0
2)	Electricity a)	**9.76E-05**	**1.29E-07**	**1.41E-07**	**4.74E-08**	**8.52E-05**	**9.87E-08**	**4.62E-08**	**3.56E-06**	**3.26E-06**	**6.22E-07**	**1.96E-06**	**2.49E-05**
	Electricity b)	**9.34E-05**	**1.23E-07**	**1.35E-07**	**4.54E-08**	**8.15E-05**	**9.44E-08**	**4.42E-08**	**3.40E-06**	**3.12E-06**	**5.95E-07**	**1.87E-06**	**2.39E-05**



1) Emissions occur during the process. 2) Emissions occur during the material production. a) Derives from crane. b) Derives from angle grinder.

Table C.4: Characterisation results for the Shredding per kg DFG_2

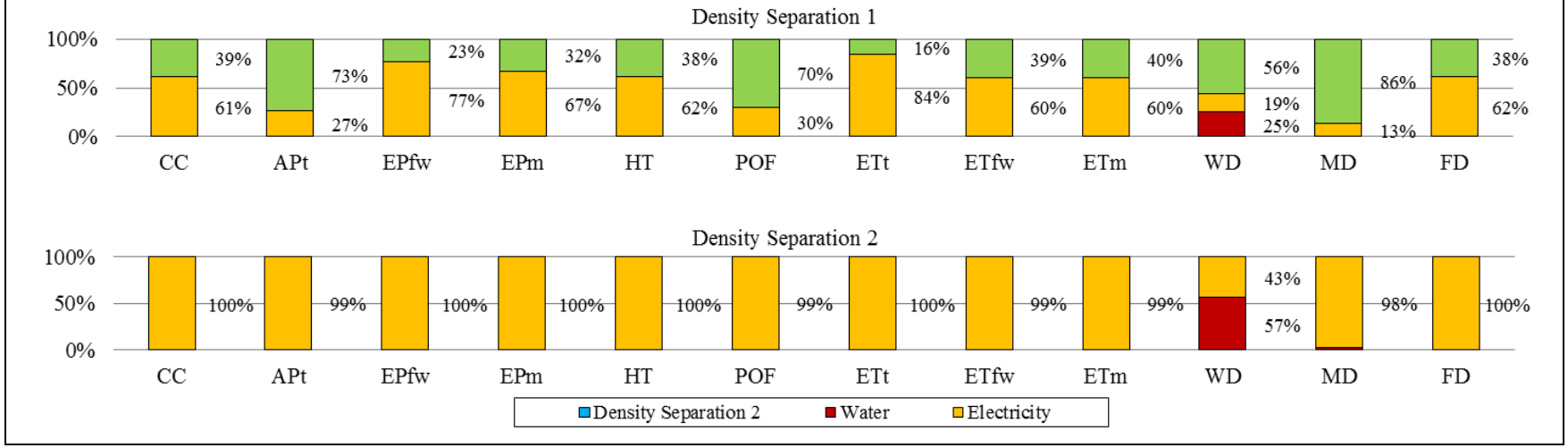
	Impact category	CC	APt	EPfw	EPm	HT	POF	ETt	ETfw	ETm	WD	MD	FD
	Unit	kg CO_2 eq	kg SO_2 eq	kg P eq	kg N eq	kg 14-DB eq	kg NMVOC			kg 14-DB	m3	kg Fe eq	kg oil eq
	Shredding	0	0	0	0	0	0	0	0	0	0	0	0
1)	Electricity a)	**1.33E-02**	**1.75E-05**	**1.91E-05**	**6.44E-06**	**1.16E-02**	**1.34E-05**	**6.27E-06**	**4.83E-04**	**4.43E-04**	**8.45E-05**	**2.66E-04**	**3.39E-03**
2)	Electricity b)	**3.42E-02**	**4.52E-05**	**4.93E-05**	**1.66E-05**	**2.99E-02**	**3.46E-05**	**1.62E-05**	**1.25E-03**	**1.14E-03**	**2.18E-04**	**6.86E-04**	**8.74E-03**



1) Emissions occur during the process. 2) Emissions occur during the energy production. a) Derives from rough shredding. b) Derives from fine shredding.

Table C.5: Characterisation results for the Density Separation per kg DFG_3

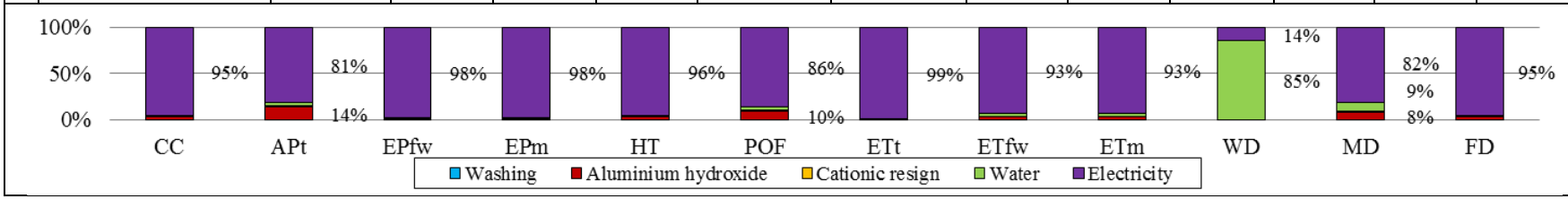
Impact category		CC	APt	EPfw	EPm	HT	POF	ETt	ETfw	ETm	WD	MD	FD
Unit		kg CO_2 eq	kg SO_2 eq	kg P eq	kg N eq	kg 14-DB eq	kg NMVOC	kg 14-DB eq			m3	kg Fe eq	kg oil eq
[1]	Density Separation	0	0	0	0	0	0	0	0	0	0	0	0
[2]	Water	1.10E-04	4.99E-07	7.52E-08	3.20E-08	8.46E-05	3.40E-07	1.61E-08	1.27E-05	1.12E-05	**3.04E-04**	1.79E-05	2.90E-05
	Electricity	**3.62E-02**	**4.78E-05**	**5.22E-05**	**1.76E-05**	**3.16E-02**	**3.66E-05**	**1.71E-05**	**1.32E-03**	**1.21E-03**	**2.31E-04**	**7.26E-04**	**9.24E-03**
	Sodium chloride [a]	**2.27E-02**	**1.30E-04**	**1.55E-05**	**8.44E-06**	**1.96E-02**	**8.44E-05**	**3.18E-06**	**8.63E-04**	**8.10E-04**	**6.70E-04**	**4.66E-03**	**5.75E-03**



[1] Emissions occur during the process. [2] Emissions occur during the material/energy production. [a] Only needed in density separation 1.

Table C.6: Characterisation results for the Washing per kg DFG_5

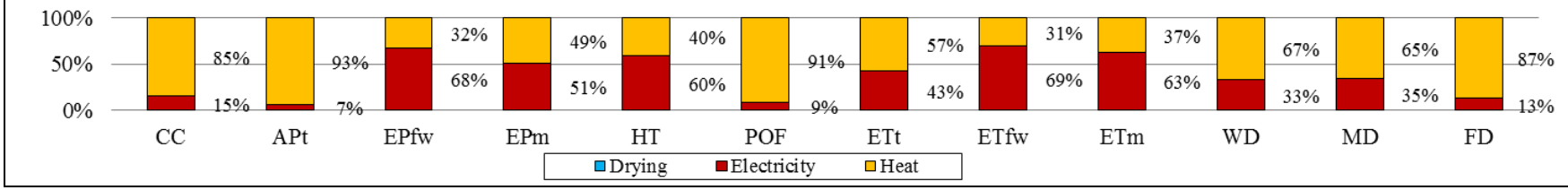
	Impact category	CC	APt	EPfw	EPm	HT	POF	ETt	ETfw	ETm	WD	MD	FD
	Unit	kg CO_2 eq	kg SO_2 eq	kg P eq	kg N eq	kg 14-DB eq	kg NMVOC	kg 14-DB eq			m3	kg Fe eq	kg oil eq
1)	Washing	0	0	0	0	0	0	0	0	0	0	0	0
2)	Aluminium hydroxide	1.67E-03	**1.19E-05**	8.35E-07	2.85E-07	1.38E-03	**5.97E-06**	1.29E-07	6.22E-05	6.18E-05	8.41E-06	**1.09E-04**	4.08E-04
	Cationic resign	1.61E-04	7.35E-07	3.10E-08	2.27E-08	3.99E-05	5.19E-07	1.02E-08	1.72E-06	1.58E-06	2.63E-06	8.03E-06	7.53E-05
	Water	7.33E-04	3.33E-06	5.01E-07	2.13E-07	5.64E-04	2.27E-06	1.07E-07	8.49E-05	7.46E-05	**2.02E-03**	**1.19E-04**	1.94E-04
	Electricity	**5.30E-02**	**7.01E-05**	**7.65E-05**	**2.58E-05**	**4.63E-02**	**5.36E-05**	**2.51E-05**	**1.93E-03**	**1.77E-03**	**3.38E-04**	**1.06E-03**	**1.36E-02**



[1] Emissions occur during the process. [2] Emissions occur during the material/energy production.

Table C.7: Characterisation results for the Drying per kg DFG_6

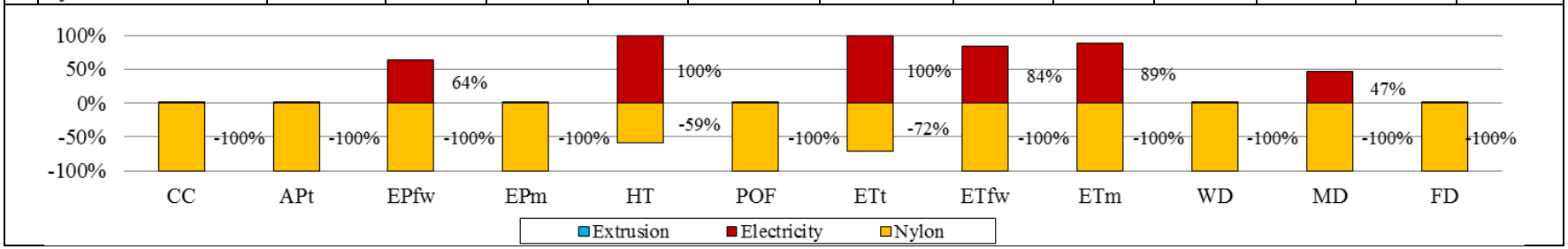
	Impact category	CC	APt	EPfw	EPm	HT	POF	ETt	ETfw	ETm	WD	MD	FD
	Unit	kg CO_2 eq	kg SO_2 eq	kg P eq	kg N eq	kg 14-DB eq	kg NMVOC	kg 14-DB eq			m3	kg Fe eq	kg oil eq
1)	Drying	0	0	0	0	0	0	0	0	0	0	0	0
2)	Electricity	**1.51E-02**	**1.99E-05**	**2.17E-05**	**7.32E-06**	**1.31E-02**	**1.52E-05**	**7.13E-06**	**5.49E-04**	**5.04E-04**	**9.60E-05**	**3.02E-04**	**3.85E-03**
	Heat	**8.31E-02**	**2.68E-04**	**1.03E-05**	**7.02E-06**	**8.92E-03**	**1.52E-04**	**9.60E-06**	**2.45E-04**	**2.93E-04**	**1.94E-04**	**5.60E-04**	**2.60E-02**



[1] Emissions occur during the process. [2] Emissions occur during the material/energy production.

Table C.8: Characterisation results for Extrusion per kg DFG_7

	Impact category	CC	APt	EPfw	EPm	HT	POF	ETt	ETfw	ETm	WD	MD	FD
	Unit	kg CO_2 eq	kg SO_2 eq	kg P eq	kg N eq	kg 14-DB eq	kg NMVOC	kg 14-DB eq			m3	kg Fe eq	kg oil eq
1)	Extrusion	0	0	0	0	0	0	0	0	0	0	0	0
2)	Electricity	1.97E-01	2.61E-04	**2.85E-04**	9.59E-05	**1.72E-01**	2.00E-04	**9.34E-05**	**7.19E-03**	**6.60E-03**	1.26E-03	**3.96E-03**	5.04E-02
	Nylon	**7.94E+00**	**2.71E-02**	**4.46E-04**	**8.98E-03**	**1.02E-01**	**2.07E-02**	**6.68E-05**	**8.52E-03**	**7.41E-03**	**2.22E-01**	**8.44E-03**	**2.66E+00**

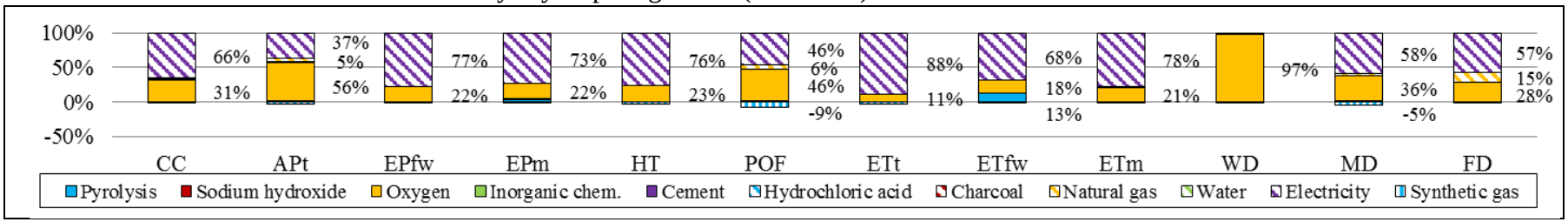


[1] Emissions occur during the process. [2] Emissions occur during the material/energy production.

Table C.9: Characterisation results for the Pyrolysis per kg DFG_3

	Impact category	CC	APt	EPfw	EPm	HT	POF	ETt	ETfw	ETm	WD	MD	FD
	Unit	kg CO_2 eq	kg SO_2 eq	kg P eq	kg N eq	kg 14-DB eq	kg NMVOC	kg 14-DB eq			m3	kg Fe eq	kg oil eq
1)	Pyrolysis	0	0	0	0	0	0	0	0	0	0	0	0
2)	Sodium hydroxide	7.28E-03	3.60E-05	4.35E-06	1.87E-06	4.14E-03	2.12E-05	7.66E-07	**1.42E-04**	1.33E-04	2.55E-04	4.53E-04	1.84E-03
	Oxygen	**3.45E-01**	**1.47E-03**	**3.00E-04**	**1.07E-04**	**1.93E-01**	**7.39E-04**	**4.24E-05**	**6.96E-03**	**6.49E-03**	**1.87E-01**	**9.06E-03**	**9.23E-02**
	Inorganic chemicals	2.24E-03	1.45E-05	9.03E-07	5.23E-07	9.71E-04	6.72E-06	2.92E-07	3.75E-05	3.61E-05	8.48E-05	1.45E-04	5.45E-04
	Cement	5.02E-04	1.04E-06	5.04E-08	4.71E-08	5.49E-05	1.00E-06	1.81E-08	1.37E-06	1.35E-06	1.17E-06	5.72E-06	5.06E-05
	Hydrochloric acid	2.80E-04	1.62E-06	1.98E-07	8.28E-08	2.92E-04	9.41E-07	1.73E-07	7.91E-06	8.44E-06	9.63E-06	3.48E-05	1.00E-04
	Charcoal	1.02E-05	1.84E-08	7.17E-10	8.93E-10	8.62E-07	1.28E-07	8.39E-10	1.05E-07	9.52E-08	1.50E-07	1.80E-07	1.22E-06
	Natural gas	2.57E-02	**1.21E-04**	6.35E-06	2.91E-06	4.68E-03	**9.70E-05**	9.49E-07	1.75E-04	1.79E-04	2.20E-04	8.39E-04	**4.89E-02**
	Water	5.46E-06	2.48E-08	3.74E-09	1.59E-09	4.20E-06	1.69E-08	8.00E-10	6.32E-07	5.56E-07	1.51E-05	8.90E-07	1.44E-06
	Electricity	**7.37E-01**	**9.74E-04**	**1.06E-03**	**3.58E-04**	**6.43E-01**	**7.45E-04**	**3.49E-04**	**2.68E-02**	**2.46E-02**	4.70E-03	**1.48E-02**	**1.88E-01**
3)	Synthetic gas	1.56E-02	8.86E-05	1.21E-05	6.15E-06	2.30E-02	**1.37E-04**	1.02E-05	5.37E-04	4.95E-04	4.15E-04	**1.21E-03**	5.52E-03

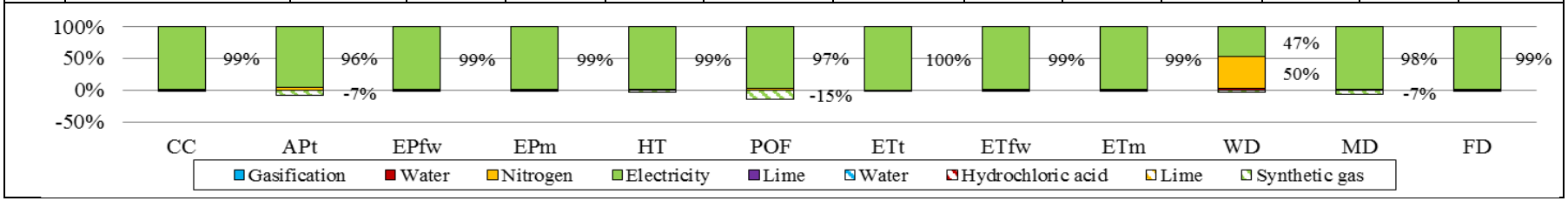
Table C.8: Characterisation results for the Pyrolysis per kg DFG_3 (continued)



[1] Emissions occur during the process. [2] Emissions occur during the material/energy production. [3] Emissions occur during the avoided material/energy production.

Table C.10: Characterisation results for the Gasification per kg DFG_3

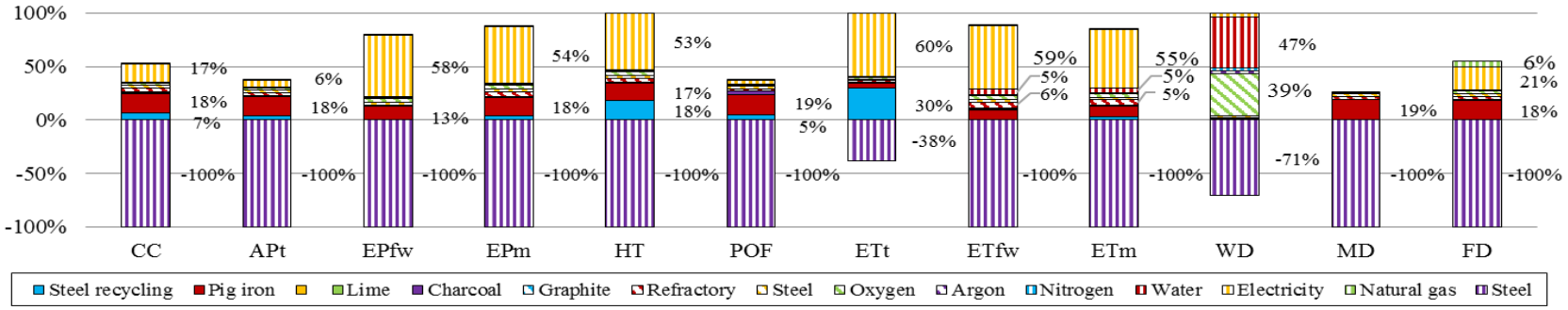
Impact category		CC	APt	EPfw	EPm	HT	POF	ETt	ETfw	ETm	WD	MD	FD
Unit		kg CO_2 eq	kg SO_2 eq	kg P eq	kg N eq	kg 14-DB eq	kg NMVOC	kg 14-DB eq			m3	kg Fe eq	kg oil eq
[1]	Gasification	0	0	0	0	0	0	0	0	0	0	0	0
[2]	Water	1.22E-04	5.54E-07	8.35E-08	3.55E-08	9.39E-05	3.77E-07	1.79E-08	1.41E-05	1.24E-05	3.37E-04	1.99E-05	3.22E-05
	Nitrogen	1.09E-02	4.67E-05	9.47E-06	3.40E-06	6.11E-03	2.39E-05	1.38E-06	2.20E-04	2.05E-04	**5.89E-03**	2.92E-04	2.94E-03
	Electricity	**8.75E-01**	**1.16E-03**	**1.26E-03**	**4.25E-04**	**7.64E-01**	**8.85E-04**	**4.14E-04**	**3.19E-02**	**2.93E-02**	**5.58E-03**	**1.76E-02**	**2.24E-01**
	Lime	1.65E-04	9.29E-07	4.77E-08	3.38E-08	4.23E-05	6.63E-07	1.39E-08	1.34E-06	1.24E-06	4.28E-06	6.21E-06	4.43E-05
	Water	4.58E-06	2.08E-08	3.13E-09	1.33E-09	3.53E-06	1.42E-08	6.71E-10	5.30E-07	4.66E-07	1.27E-05	7.46E-07	1.21E-06
	Hydrochloric acid	7.81E-07	4.52E-09	5.50E-10	2.31E-10	8.13E-07	2.62E-09	4.81E-10	2.20E-08	2.35E-08	2.68E-08	9.71E-08	2.80E-07
	Lime	3.13E-07	1.77E-09	9.07E-11	6.43E-11	8.03E-08	1.26E-09	2.64E-11	2.55E-09	2.37E-09	8.14E-09	1.18E-08	8.42E-08
[3]	Synthetic gas	1.56E-02	**8.86E-05**	1.21E-05	6.15E-06	2.30E-02	**1.37E-04**	1.02E-05	5.37E-04	4.95E-04	4.15E-04	**1.21E-03**	5.52E-03



[1] Emissions occur during the process. [2] Emissions occur during the material/energy production. [3] Emissions occur during the avoided material/energy production.

Table C.11: Characterisation results for the Steel recycling per kg scrap steel

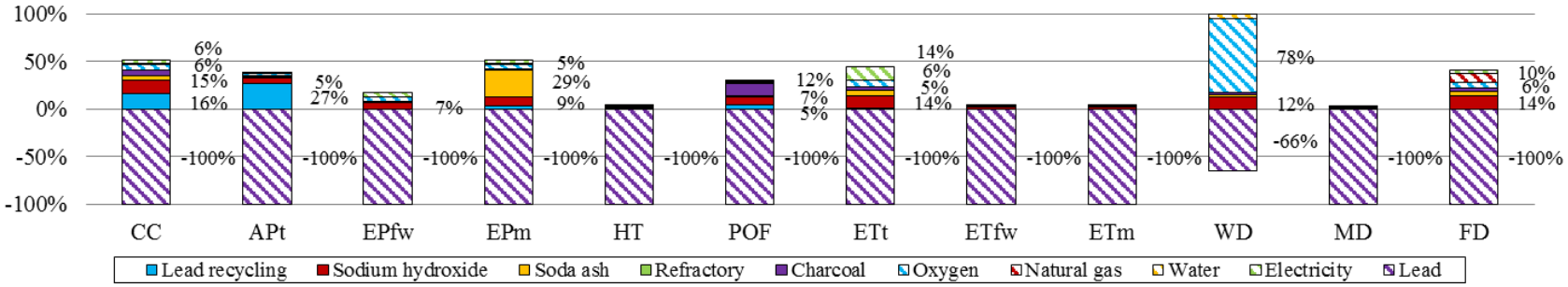
Impact category		CC	APt	EPfw	EPm	HT	POF	ETt	ETfw	ETm	WD	MD	FD
Unit		kg CO_2 eq	kg SO_2 eq	kg P eq	kg N eq	kg 14-DB eq	kg NMVOC	kg 14-DB eq			m3	kg Fe eq	kg oil eq
1)	Steel recycling	**1.16E-01**	2.30E-04	0	8.85E-06	**8.33E-02**	**3.36E-04**	**6.54E-05**	6.85E-06	4.78E-04	0	0	0
	Pig iron	**1.23E-01**	**4.70E-04**	**3.64E-05**	**1.86E-05**	**3.17E-02**	**5.77E-04**	4.43E-06	6.91E-04	6.88E-04	2.25E-04	**8.78E-02**	**2.58E-02**
	Pig iron	**1.73E-01**	**6.58E-04**	**5.09E-05**	**2.60E-05**	**4.44E-02**	**8.08E-04**	6.19E-06	**9.68E-04**	**9.63E-04**	3.15E-04	**1.23E-01**	**3.61E-02**
	Lime	3.10E-03	1.75E-05	9.00E-07	6.38E-07	7.97E-04	1.25E-05	2.62E-07	2.53E-05	2.35E-05	8.07E-05	1.17E-04	8.36E-04
	Charcoal	2.15E-02	3.87E-05	1.51E-06	1.87E-06	1.81E-03	2.68E-04	1.76E-06	2.21E-04	2.00E-04	3.14E-04	3.79E-04	2.56E-03
	Graphite	2.73E-04	1.81E-06	8.55E-08	7.77E-08	8.69E-05	1.68E-06	5.06E-08	2.71E-06	2.81E-06	1.62E-06	1.43E-05	8.01E-05
	Refractory	5.58E-02	1.53E-04	5.85E-06	9.28E-06	1.57E-02	1.52E-04	3.06E-06	**9.45E-04**	**8.17E-04**	1.34E-04	2.97E-02	9.54E-03
2)	Steel	4.41E-02	1.67E-04	1.86E-05	6.72E-06	1.23E-02	1.96E-04	2.26E-06	4.58E-04	4.52E-04	8.47E-04	2.93E-02	9.04E-03
	Oxygen	3.19E-02	1.36E-04	2.78E-05	9.93E-06	1.79E-02	6.84E-05	3.93E-06	6.44E-04	6.01E-04	**1.73E-02**	8.39E-04	8.55E-03
	Argon	3.59E-03	1.63E-05	2.13E-06	8.21E-07	1.40E-03	8.77E-06	2.22E-07	4.95E-05	4.54E-05	1.30E-03	5.21E-05	9.05E-04
	Nitrogen	2.05E-03	8.74E-06	1.77E-06	6.36E-07	1.14E-03	4.47E-06	2.57E-07	4.11E-05	3.84E-05	1.10E-03	5.47E-05	5.50E-04
	Water	7.70E-03	3.50E-05	5.27E-06	2.24E-06	5.92E-03	2.38E-05	1.13E-06	**8.91E-04**	**7.83E-04**	**2.13E-02**	1.25E-03	2.03E-03
	Electricity	**2.78E-01**	**3.68E-04**	**4.01E-04**	**1.35E-04**	**2.43E-01**	2.81E-04	**1.32E-04**	**1.01E-02**	**9.29E-03**	1.77E-03	5.58E-03	**7.10E-02**
	Natural gas	1.02E-02	4.79E-05	2.51E-06	1.15E-06	1.85E-03	3.84E-05	3.76E-07	6.92E-05	7.06E-05	8.72E-05	3.32E-04	**1.94E-02**
3)	Steel	**1.65E+00**	**6.24E-03**	**6.95E-04**	**2.52E-04**	**4.59E-01**	**7.34E-03**	**8.45E-05**	**1.71E-02**	**1.69E-02**	**3.17E-02**	**1.10E+0**	**3.38E-01**



[1] Emissions occur during the process. [2] Emissions occur during the material/energy production. [3] Emissions occur during the avoided material/energy production.

Table C.12: Characterisation results for the Lead recycling per kg scrap lead

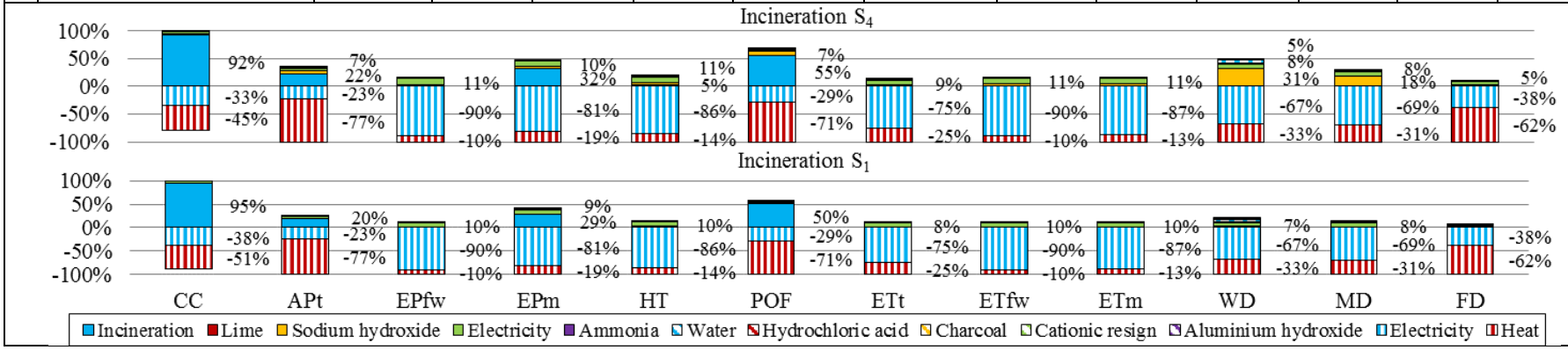
Impact category		CC	APt	EPfw	EPm	HT	POF	ETt	ETfw	ETm	WD	MD	FD
Unit		kg CO_2 eq	kg SO_2 eq	kg P eq	kg N eq	kg 14-DB eq	kg NMVOC	kg 14-DB eq			m3	kg Fe eq	kg oil eq
1)	Lead recycling	**2.16E-01**	**5.00E-03**	1.22E-07	1.95E-05	3.79E-02	**4.06E-04**	8.84E-07	8.33E-05	9.04E-05	0	0	0
2)	Sodium hydroxide	**2.02E-01**	**1.00E-03**	**1.21E-04**	**5.21E-05**	1.15E-01	**5.90E-04**	**2.13E-05**	3.95E-03	3.69E-03	**7.10E-03**	1.26E-02	**5.12E-02**
	Soda ash	5.22E-02	2.76E-04	2.04E-05	**1.60E-04**	2.55E-02	1.40E-04	**8.52E-06**	1.07E-03	9.96E-04	1.41E-03	5.26E-03	1.59E-02
	Refractory	1.49E-03	4.07E-06	1.56E-07	2.48E-07	4.19E-04	4.05E-06	8.15E-08	2.52E-05	2.18E-05	3.59E-06	7.93E-04	2.55E-04
	Charcoal	**8.14E-02**	1.47E-04	5.72E-06	7.12E-06	6.87E-03	**1.02E-03**	6.69E-06	8.39E-04	7.59E-04	1.19E-03	1.44E-03	9.71E-03
	Oxygen	**8.25E-02**	3.52E-04	7.18E-05	**2.57E-05**	4.62E-02	1.77E-04	**1.02E-05**	1.66E-03	1.55E-03	**4.48E-02**	2.17E-03	**2.21E-02**
	Natural gas	1.78E-02	8.39E-05	4.40E-06	2.02E-06	3.24E-03	6.73E-05	6.58E-07	1.21E-04	1.24E-04	1.53E-04	5.82E-04	**3.39E-02**
	Water	8.94E-04	4.06E-06	6.12E-07	2.60E-07	6.88E-04	2.76E-06	1.31E-07	1.04E-04	9.10E-05	2.47E-03	1.46E-04	2.36E-04
	Electricity	4.69E-02	6.20E-05	6.76E-05	2.28E-05	4.09E-02	4.74E-05	**2.22E-05**	1.71E-03	1.57E-03	2.99E-04	9.41E-04	1.20E-02
3)	Lead	**1.36E+00**	**1.83E-02**	**1.71E-03**	**5.57E-04**	**6.32E+00**	**8.14E-03**	**1.58E-04**	**2.26E-01**	**1.99E-01**	**3.77E-02**	**1.33E+0**	**3.54E-01**



[1] Emissions occur during the process. [2] Emissions occur during the material/energy production. [3] Emissions occur during the avoided material/energy production.

Table C.13: Characterisation results for the Incineration per kg input

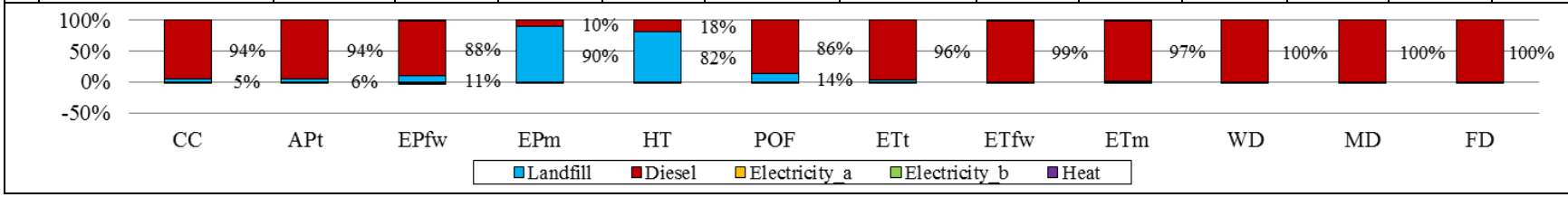
Impact category		CC	APt	EPfw	EPm	HT	POF	ETt	ETfw	ETm	WD	MD	FD
Unit		kg CO_2 eq	kg SO_2 eq	kg P eq	kg N eq	kg 14-DB eq	kg NMVOC	kg 14-DB eq			m3	kg Fe eq	kg oil eq
1)	Incineration	**1.55E+00**	**6.94E-04**	0	**1.08E-04**	1.23E-02	**1.09E-03**	2.92E-06	1.19E-05	2.82E-05	0	0	0
2)	Lime [a)]	1.07E-03	6.03E-06	3.10E-07	2.20E-07	2.75E-04	4.31E-06	9.03E-08	8.71E-06	8.09E-06	2.78E-05	4.03E-05	2.88E-04
	Lime [b)]	1.09E-04	6.15E-07	3.16E-08	2.24E-08	2.80E-05	4.39E-07	9.20E-09	8.87E-07	8.24E-07	2.83E-06	4.11E-06	2.93E-05
	Sodium hydroxide [a)]	4.73E-02	**2.34E-04**	2.83E-05	1.22E-05	**2.69E-02**	**1.38E-04**	4.98E-06	9.23E-04	8.63E-04	1.66E-03	2.95E-03	1.20E-02
	Sodium hydroxide [b)]	4.83E-03	2.39E-05	2.89E-06	1.24E-06	2.75E-03	1.41E-05	5.08E-07	9.42E-05	8.81E-05	**1.69E-04**	**3.01E-04**	1.22E-03
	Electricity	6.90E-02	9.12E-05	**9.95E-05**	**3.35E-05**	**6.02E-02**	6.97E-05	**3.26E-05**	**2.51E-03**	**2.30E-03**	**4.39E-04**	**1.38E-03**	**1.76E-02**
	Ammonia	7.20E-03	4.37E-05	5.82E-07	7.05E-07	1.79E-03	1.38E-05	2.09E-06	3.34E-05	5.14E-05	1.55E-04	1.69E-04	2.78E-03
	Water	9.16E-05	4.16E-07	6.27E-08	2.66E-08	7.05E-05	2.83E-07	1.34E-08	1.06E-05	9.32E-06	**2.53E-04**	1.49E-05	2.42E-05
	Hydrochloric acid	9.41E-05	5.45E-07	6.63E-08	2.78E-08	9.79E-05	3.16E-07	5.80E-08	2.65E-06	2.83E-06	3.23E-06	1.17E-05	3.37E-05
	Charcoal	1.09E-03	1.97E-06	7.68E-08	9.56E-08	9.24E-05	1.37E-05	8.99E-08	1.13E-05	1.02E-05	1.60E-05	1.93E-05	1.31E-04
	Cationic resign	3.60E-03	1.64E-05	6.91E-07	5.07E-07	8.89E-04	1.16E-05	2.27E-07	3.84E-05	3.52E-05	5.87E-05	1.79E-04	1.68E-03
	Aluminium hydr.	1.03E-04	7.37E-07	5.18E-08	1.77E-08	8.58E-05	3.70E-07	7.98E-09	3.85E-06	3.83E-06	5.21E-07	6.77E-06	2.53E-05
3)	Electricity [a)]	**5.62E-01**	**7.44E-04**	**8.11E-04**	**2.73E-04**	**4.91E-01**	**5.69E-04**	**2.66E-04**	**2.05E-02**	**1.88E-02**	**3.58E-03**	**1.13E-02**	**1.44E-01**
	Electricity [b)]	**6.15E-01**	**8.14E-04**	**8.87E-04**	**2.99E-04**	**5.37E-01**	**6.22E-04**	**2.91E-04**	**2.24E-02**	**2.06E-02**	**3.92E-03**	**1.23E-02**	**1.57E-01**
	Heat [a)]	**7.64E-01**	**2.47E-03**	**9.43E-05**	**6.45E-05**	**8.19E-02**	**1.40E-03**	**8.82E-05**	**2.25E-03**	**2.69E-03**	**1.78E-03**	**5.14E-03**	**2.39E-01**
	Heat [b)]	**8.37E-01**	**2.70E-03**	**1.03E-04**	**7.07E-05**	**8.98E-02**	**1.53E-03**	**9.67E-05**	**2.46E-03**	**2.95E-03**	**1.95E-03**	**5.64E-03**	**2.62E-01**



[1)] Process Emissions. [2)] Emissions from material/energy production. [3)] Emissions from avoided material/energy production. [a)] Refers to scenario 1. [b)] Refers to scenario 4.

Table C.14: Characterisation results for the Landfill per kg DFG_1

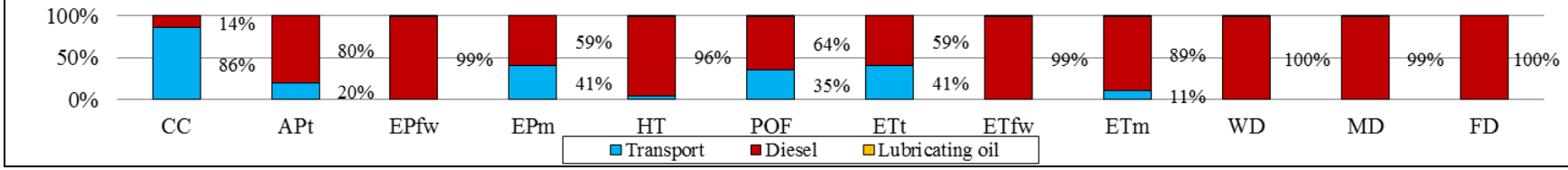
Impact category		CC	APt	EPfw	EPm	HT	POF	ETt	ETfw	ETm	WD	MD	FD
Unit		kg CO_2 eq	kg SO_2 eq	kg P eq	kg N eq	kg 14-DB eq	kg NMVOC	kg 14-DB eq			m3	kg Fe eq	kg oil eq
1)	Landfill	**2.77E-02**	**2.81E-04**	**7.35E-06**	**9.27E-04**	**3.59E-01**	**4.80E-04**	**2.25E-06**	2.10E-05	5.54E-05	0	0	0
2)	Diesel	**4.76E-01**	**4.38E-03**	**5.91E-05**	**9.77E-05**	**7.78E-02**	**2.99E-03**	**5.57E-05**	**3.17E-03**	**2.57E-03**	**5.70E-03**	**1.50E-02**	**1.13E+00**
	Electricity_a	3.68E-04	4.87E-07	5.31E-07	1.79E-07	3.21E-04	3.72E-07	1.74E-07	1.34E-05	1.23E-05	2.35E-06	7.38E-06	9.40E-05
3)	Electricity_b	3.74E-04	4.95E-07	5.39E-07	1.82E-07	3.26E-04	3.78E-07	1.77E-07	1.36E-05	1.25E-05	2.38E-06	7.50E-06	9.55E-05
	Heat	1.02E-04	3.28E-07	1.25E-08	8.58E-09	1.09E-05	1.86E-07	1.17E-08	2.99E-07	3.57E-07	2.37E-07	6.84E-07	3.17E-05



[1] Process Emissions. [2] Emissions from material/energy production. [3] Emissions from avoided material/energy production.

Table C.15: Characterisation results for the Transport per kg input

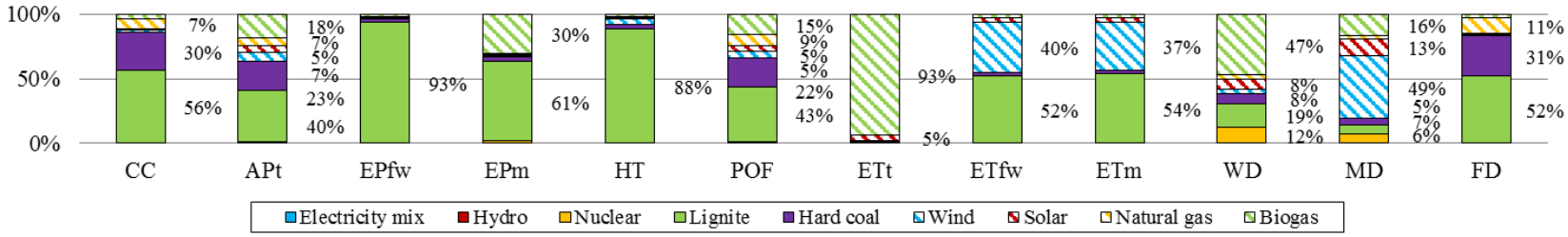
Impact category		CC	APt	EPfw	EPm	HT	POF	ETt	ETfw	ETm	WD	MD	FD
Unit		kg CO_2 eq	kg SO_2 eq	kg P eq	kg N eq	kg 14-DB eq	kg NMVOC	kg 14-DB eq			m3	kg Fe eq	kg oil eq
1)	Transport	**3.33E-01**	**1.26E-04**	0	**8.02E-06**	3.58E-04	**1.92E-04**	**4.47E-06**	5.93E-07	**3.61E-05**	0	0	0
2)	Diesel	**5.58E-02**	**5.13E-04**	**6.93E-06**	**1.15E-05**	**9.11E-03**	**3.50E-04**	**6.52E-06**	**3.71E-04**	**3.01E-04**	**6.68E-04**	**1.76E-03**	**1.32E-01**
	Lubricating oil	1.75E-04	1.35E-06	5.11E-08	3.72E-08	6.00E-05	2.47E-06	1.88E-08	2.57E-06	2.29E-06	1.84E-06	1.23E-05	2.88E-04



[1] Process Emissions. [2] Emissions from material/energy production.

Table C.16: Electricity mix - Characterisation results for generating 1 kWh

Impact category		CC	APt	EPfw	EPm	HT	POF	ETt	ETfw	ETm	WD	MD	FD
Unit		kg CO_2 eq	kg SO_2 eq	kg P eq	kg N eq	kg 14-DB eq	kg NMVOC	kg 14-DB eq			m3	kg Fe eq	kg oil eq
[1]	Electricity mix	0	0	0	0	0	0	0	0	0	0	0	0
[2]	Hydro	4.54E-05	1.79E-07	1.63E-08	9.19E-09	2.72E-05	2.05E-07	3.93E-09	6.18E-06	5.42E-06	6.67E-07	2.15E-05	9.23E-06
	Nuclear	3.96E-04	1.99E-06	2.69E-07	1.45E-06	9.31E-04	1.64E-06	1.05E-07	1.56E-05	1.67E-05	**1.14E-04**	**1.88E-04**	1.00E-04
	Lignite	**8.32E-02**	**7.77E-05**	**1.99E-04**	**4.38E-05**	**1.13E-01**	6.39E-05	4.12E-07	**2.77E-03**	**2.65E-03**	**1.76E-04**	**2.07E-04**	**1.96E-02**
	Hard coal	**4.38E-02**	**4.42E-05**	6.28E-06	2.71E-06	5.03E-03	3.26E-05	3.46E-07	1.34E-04	1.29E-04	**7.48E-05**	**1.51E-04**	**1.16E-02**
	Wind	1.55E-03	**1.29E-05**	2.52E-06	8.22E-07	5.13E-03	7.53E-06	3.91E-07	**2.13E-03**	**1.84E-03**	3.06E-05	**1.45E-03**	3.98E-04
	Solar	1.88E-03	**1.01E-05**	1.22E-06	7.33E-07	1.78E-03	7.21E-06	**3.35E-06**	1.51E-04	1.43E-04	**7.24E-05**	**3.76E-04**	4.87E-04
	Natural gas	**1.08E-02**	**1.31E-05**	5.72E-07	4.20E-07	4.44E-04	1.31E-05	8.86E-08	1.62E-05	1.66E-05	3.23E-05	9.26E-05	**4.27E-03**
	Biogas	5.64E-03	**3.47E-05**	3.08E-06	**2.16E-05**	2.49E-03	2.29E-05	**6.50E-05**	1.39E-04	1.21E-04	**4.38E-04**	**4.74E-04**	1.13E-03



[1] Process Emissions. [2] Emissions from material/energy production.